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A study of phosphorus loading and water quality implications was conducted for the Oregon coastal lakes. The study was based on existing data for lake total phosphorus concentrations and for watershed land uses. A phosphorus mass-balance model was developed to predict lake total phosphorus concentrations from estimated phosphorus loading from land uses within the lake's watershed. Uncertainty in total phosphorus concentration estimates are included in the model, and model predictions are considered to be moderately to highly reliable.

The Oregon coastal lake phosphorus mass-balance model was calibrated from data for 12 Oregon coastal lakes. Land use phosphorus loading coefficients for forestry, the coastal dunal aquifer, and precipitation were derived from data specific to the Oregon coastal region, while other phosphorus loading coefficients were estimated based on correlations between literature values and Oregon coastal conditions.

The model may be used as an aid for land use management decisions by estimating water quality effects of projected land use changes. A case study of Mercer Lake was used to illustrate the model application.

Oregon Coastal Lake Study:
Phosphorus Loading and Water Quality Implications

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OCLS = Oregon Coastal Lake Study

TP = total phosphorus

P = phosphorus

A = lake surface area (km^2)

$\bar{z}$ = lake mean depth (m)

RO = average annual runoff (m/yr)

ρ = lake flushing rate (yr^{-1})

R = sediment P retention factor; dimensionless

S = lake sensitivity coefficient

q_s = lake surface overflow rate (m/yr)

WSA = lake watershed area (km^2)

WSA subscripts: bg, ag, urb, bog, and res denote forest (background), agricultural, urban, wetland/bog, and residential, respectively.

UP = phosphorus loading from upgradient lake

D.U. = nearshore dwelling unit on septic system

FORY = P-loading from forested area . . . . ($kg/km^2/yr$)

PREL = " " precipitation and atmospheric inputs . . . . "

DUNL = " " dunal areas . . . . "

RESL = " " residential areas . . . . "

RANGL = " " agricultural/rangeland areas . . . . "

URBL = " " urban areas . . . . "

WETL = " " wetland/bog areas . . . . "

SEPL = " " nearshore septic system. . . . . (kg/D.U./yr)

Atlas = Johnson et al. (1985) = Atlas of Oregon Lakes

OREGON COASTAL LAKE STUDY:
PHOSPHORUS LOADING AND WATER QUALITY IMPLICATIONS

CHAPTER 1: INTRODUCTION AND OBJECTIVES

Coastal lakes in Oregon play an integral part of the total water resource system, and some are vitally important to local economies. Their attractiveness may be viewed in an array of recreational uses such as swimming, boating, fisheries and wildlife, tourism, and camping; or they may be perceived as prime areas for residential development. Other interests may lie strictly in water quality alone, which is the case with Clear Lake, the water source for the city of Florence. Without proper management of activities affecting these lakes, the possibility exists that they could be adversely altered (Johnson, et al., 1985).

In the last several decades, increases in our wealth and leisure time have created greater demands on our natural and recreational resources. Many lakes have been the recipient of this increased activity, either directly or inadvertently, and some are showing adverse effects. Lakes readily accessible to population centers tend to receive the greatest direct growth pressures from residential developments and recreational activities.

Water quality deterioration in lakes can be caused by many different factors. Unfortunately, those lakes which are feeling population pressures and intensified land use activities within their watersheds will usually experience internal chemical and biological changes. These changes occur in the form of accelerated biological activity such as aquatic plant and algal growth, which is transferred through the food chain and eventually affects the chemical interactions in the lake as well. These changes, called cultural eutrophication of lakes, are caused from increases in the nutrients nitrogen and phosphorus. The increase in nutrient load is primarily derived upgradient in the watershed, and stems from residential/commercial, agricultural, and forest practice activities (Gilliom, 1982). Increases in lake fertilization are realized from mechanisms such as sewage treatment and industrial effluent, septic tank leachate, lawn fertilization, agricultural runoff (Reckhow and Simpson, 1980; Lee and Rast, 1978), silvicultural runoff (Harr and Fredriksen, 1988), and general erosion. Increased nutrient input can also be realized in the form of precipitation (Gilliom, 1982), where windblown soil containing nutrients are incorporated into atmospheric moisture.

Limnological studies are based upon very complex and interdependent physical, chemical, and biological interactions within a lake. Although there are many ways to classify lakes, trophic status is a generally accepted method, where they are categorized according to biological productivity. An unproductive lake would be considered oligotrophic, and a highly productive lake would be eutrophic. Biological productivity and speciation is dependent upon the physical and chemical characteristics of the lake, while the physical and chemical properties can be altered due to biological activity (Wetzel, 1983; Johnson, et al., 1985).

Some of the parameters used in limnological studies which effect the physical, chemical, and biological activities are depth, surface area, hydraulic retention time, mixing capabilities, geomorphology of the basin and watershed, secci depth, chlorophyll-a, oxygen depletion, and thermal stratification. These measures, along with nutrients supplied from the drainage basin and the climate, can determine the dynamics of the biological population.

There is widespread agreement that phosphorus is most often growth-limiting and the most controllable nutrient causing increased algal production in temperate lakes (Reckhow, 1979; Schaffner and Oglesby, 1978; Lee and Rast, 1978; Gilliom, 1984). Thus much research has focused on developing predictive models for lake trophic status and methodologies for lake restoration based on phosphorus loading to lakes.

The purpose of this study is to develop a model which estimates total phosphorus loading to at least ten Oregon coastal lakes. These calculations must include uncertainty, and would be reflective of the water quality conditions and trophic status of the lakes. The predictions, coupled with uncertainty calculations, may be used as a land use planning tool which provides insight into different land use scenarios within the watershed. The lakes of interest for this study are: Cullaby, Devils, Eckman, Triangle, Mercer, Sutton, Collard, Clear, Munsel, Cleawox, Woahink, Siltcoos, Tahkenitch, Eel, North Tenmile, Tenmile, Loon, Floras, and Garrison.

The objectives of this study were to:

1) Summarize selected water quality data for the 19 Oregon coastal lakes of interest.

2) Summarize watershed land use data for the subset of Oregon coastal lakes selected for study.

3) Derive or select phosphorus loading coefficients which best represent watershed land uses for Oregon coastal lakes.

4) Adapt a mass-balance-type phosphorus loading model to the Oregon coastal lakes including estimates of uncertainty.

5) Illustrate use of the phosphorus loading model to assess water quality of an Oregon coastal lake under different land use scenarios.

EMPIRICAL NUTRIENT LOADING MODELS

Many models for lake management are based on the assumption that the lake may be treated as a control volume (Reckhow, 1979). This "black box" approach ignores the internal mechanisms of the lake, and empirically quantifies the nutrient's interfacial transactions. The amount of material entering and leaving the lake is accounted for through mechanisms such as natural flow, precipitation, evaporation, and sedimentation. Since phosphorus is considered to be the most biologically limiting and controllable nutrient, most empirical models, and their respective coefficients, are based upon that premise.

Biffi (1963) developed a nutrient model in which the lake was assumed to be a continuously stirred tank reactor (CSTR). At steady state, the model assumed a constant supply of material to the lake, and the outflow contained a concentration equal to the lake concentration. Although Biffi's CSTR approach has been widely accepted (Dillon, 1974; Reckhow, 1979), he did not account for activity at the sediment interface, where numerous chemical and biological reactions occur. In essence, his model was conservative, not accommodating material losses through sedimentation. Nutrients such as phosphorus, nitrogen, and carbon are non-conservative (Dillon, 1974). Biffi's model also failed to consider thermal stratification, but this was included in a modification proposed by Sweers (1969), as cited by Dillon (1974).

According to Dillon (1974), Piontelli and Tonolli (1964) were the first to consider material loss to sediments in a model. Vollenweider (1964) also included sedimentation in his model, and correctly assumed that the sedimentation rate was proportional to the concentration of the substance in the water, whereas Piontelli and Tonolli (1964) wrongly assumed it to be dependent upon the influent concentration (Dillon, 1974).

Vollenweider's continued work (1968, 1969, 1973, 1975) included assumptions that have become widely accepted (Dillon, 1974; Reckhow, 1979). He concluded that phosphorus was the most common growth-limiting nutrient because carbon and nitrogen involve gas phase equilibria with the atmosphere and are generally available in excess. Vollenweider developed an empirical settling rate for phosphorus and then correlated relationships among phosphorus loading rates, hydraulic detention times, and mean depth to develop a loading diagram to predict the lake's trophic status (Lee and Rast, 1978).

Two distinct approaches for lake classification models have generally been proposed. Although much of the same information is used in the application of either type of model, they are presented in different formats.

One method classifies lakes into trophic status by directly using data from lake water samples. Measurements of total phosphorus, total nitrogen, chlorophyll-a, secci depth, and other constituents are generally needed. The second approach predicts the trophic state of a lake from phosphorus loading data, and lake geomorphology, without the requirement of in-lake measurements. The first, or trophic state criteria method is designed to provide a multivariable index of the present water quality. The second, or loading criteria approach, uses a single nutrient to predict the lake's carrying capacity and quality changes over time. It also represents a very useful tool for planning and watershed management for lake water quality (Reckhow, 1979). Since the trophic status of the Oregon coastal lakes in this particular study have already been established (Johnson et al., 1985), emphasis is placed on phosphorus loading criteria models for planning and watershed management.

Further refinements by Dillon and Rigler (1975), Dillon and Kirchner (1975), Vollenweider (1975), Larsen and Mercier (1975), and others, served as the basis for two separate comprehensive studies by Reckhow (1977) and Walker (1977), where larger data bases were incorporated into export coefficients (reducing geographical constraints on models), and prediction uncertainties were first considered (Reckhow, 1979).

Gilliom added to the empirical lake modeling efforts with his work in the Puget Sound area of Washington. His procedures were derived from the aforementioned modelers, and included phosphorus loading estimations from different watershed land uses such as: forestry; agricultural; residential; and precipitation. The breakdown of phosphorus loading from different land uses, and the magnitude of their respective export coefficients was also used by previous investigators (Dillon and Rigler, 1975; Reckhow and Simpson, 1980). Gilliom similarly incorporated uncertainty into his model as did Reckhow (1977), Walker (1977), and others.

BASIC MODEL

Vollenweider-type models are based on a mass balance of the lake's phosphorus, in the following form:

$$V\cdot\frac{dP}{dt}=L-R\cdot L-Q\cdot P \tag{1}$$

Where: P = lake phosphorus concentration, (µg/L);

L = the total phosphorus (TP) loading to the lake, (Kg/yr);

V = lake volume, ($10^6 m^3$);

Q = annual flow rate, ($10^6 m^3/yr$);

R = the lake's phosphorus retention coefficient (decimal percentage of L retained in the lake without increasing TP concentration), dimensionless.

The differences in the predictive nutrient loading models from the literature are minimal (Reckhow,1979), and most models at steady state are similar in form to that of Gilliom (1982):

$$(\bar{P})_{\infty}=\frac{L\cdot(1-R)}{\bar{z}\cdot A\cdot\rho} \tag{2}$$

Where: $(\bar{P})_{\infty}$ = the lake's mean total phosphorus (TP) concentration at steady state, in micrograms per liter (µg/L);

$\bar{z}$ = mean depth, in meters;

A = lake surface area, in square kilometers (km^2);

ρ = lake-flushing rate, in times per year that a volume of water equal to the lake's volume flows through the lake, (yr^{-1}).

This mass balance model states that the average phosphorus concentration in a lake is determined by the amount of phosphorus input to the lake, less the phosphorus amounts lost through sedimentation and outflow.

Independently, Vollenweider (1976) and Larsen and Mercier (1976) found that R is a linear function of the phosphorus settling velocity, and can be approximated by:

$$R=\frac{1}{1+\sqrt{\rho}} \tag{3}$$

The flushing rate can be estimated from:

$$\rho=\frac{WSA \cdot RO}{\bar{z} \cdot A} \tag{4}$$

Where: WSA= watershed area, (km^2); and

RO = average annual runoff, (m/yr)

If all constant values for a particular lake are combined in equation 1, it can be simplified to:

$$(\bar{P})_{\infty}=L \cdot Constant \tag{5}$$

Where the constants can be defined as the lake's sensitivity coefficient, or S, and shown as:

$$S=\frac{1-R}{\bar{z} \cdot A \cdot \rho} \tag{6}$$

Thus, equation 1 becomes:

$$(\bar{P})_{\infty}=L \cdot S \tag{7}$$

Selection of the most appropriate model must incorporate uncertainty into the output (Reckhow, 1979), and should be based on:

1. Similar conditions (geography, climate, size, depth, thermal stratification, trophic state, etc.).

2. Model derived from large data base.

3. Previous success for modeling similar lakes.

4. Model documentation (model use, misuse, limitations, etc.).

The uncertainty of phosphorus loading models can be quite large, especially when using indirect estimates of phosphorus loading such as literature export coefficients (Reckhow, 1979). Thus, the predictive information is more valuable as a decision-making tool if the precision is known, therefore uncertainty should be incorporated into lake modeling.

For Oregon's Coastal lakes it appears most appropriate to follow Gilliom's (1978,1982,1984) modeling approach, based on the following:

1. His approach is derived from the well established works of Dillon (1975), Reckhow (1979), Larsen and Mercier (1976), and others (Gilliom, 1982, 1984).

2. His approach facilitates the use of available existing data, enabling phosphorus export coefficients to be empirically derived.

3. His model contains statistical uncertainty analysis.
4. His model is calibrated for a region similar in:
    a) latitude
    b) climate
    c) proximity to the ocean
    d) land use activities.

The derivation of model uncertainty is addressed in Chapter 4.

CHAPTER 2: FATE & TRANSPORT OF PHOSPHORUS IN THE ENVIRONMENT

NATURAL SOURCES

Introduction

Phosphorus plays a vital role in all forms of life (Hooper, 1973; Wetzel, 1983; Gaudy,1988). It is an essential element in ATP (adenosine triphosphate), which is required by all energy transformation systems in living cells. Phosphorus is also an essential requirement in nucleic acids, which helps to facilitate the structural formation and growth of all cells. Thus it is easily perceived why approximately 90% of the phosphorus in fresh water systems is in some sort of organic form (Hooper, 1973; Wetzel, 1983) such as: 1) organic compounds of living and dead particulate (seston); 2) filterable organic compounds (dissolved); 3) organic compounds of macrophytes; 4) the phosphorus in free swimming animals, and; 5) phosphorus in bottom sediments.

Inorganic phosphorus in fresh water systems (approximately 10% of the total phosphorus) is composed of orthophosphates and polyphosphates (molecularly dehydrated phosphates). Polyphosphates gradually hydrolize in aqueous solution, and revert to the ortho form (where they were derived), but the rate is dependent upon temperature, pH, and enzymatic activity (Sawyer and McCarty, 1978). The orthophosphates ($H_2PO_4^-$, HPO_4^{-2}, PO_4^{-3}) are of greatest interest because they represent the form readily available for biological uptake (Hooper, 1973; Wetzel, 1983). The orthophosphates can be derived enzymatically (via organic breakdown) or brought into the aquatic system through hydrologic means. The factors affecting phosphorus input into lake systems will be investigated below.

Total phosphorus is generally partitioned into particulate and dissolved fractions. Phosphorus designated as "dissolved", is the fraction that passes through a filter which retains bacteria (0.45 μ)[1]. Orthophosphate is a major constituent of the

[1]Nomenclature and sampling techniques for orthophosphate in some past studies have created ambiguities with respect to data interpretations (Chamberlain and Shapiro, 1973;

dissolved fraction.

Although it is beyond the scope of this study to examine the enormous complexities and many unknowns involved in phosphorus chemical interactions, it is of interest to briefly consider the pathways by which phosphorus is transported in the environment.

Geology

Phosphorus is the eleventh most abundant element in the earth's crust, but is considered a trace element because it forms only about 0.1% of the rocks within the crust (Wolf, 1992; McKelvey, 1973). It occurs naturally in more than 200 minerals as a phosphate (PO_4^{-3}) compound (Wolf, 1992; McKelvey, 1973; Fisher, 1973).

Most phosphorus in the earth's crust is present as a species of the apatite group. In igneous and metamorphic rocks, the most common species is fluorapatite ($Ca_5(PO_4)_3F$), with a content generally less than 12% (Wolf, 1992). In sedimentary rocks the prominent species is carbonate fluorapatite. Because of the diversity of the elements in sedimentary rocks, along with differing biological and weathering processes, the phosphorus content is generally low (less than 0.2%). However local areas sometimes contain much higher phosphate concentrations (McKelvey, 1973).

Since phosphorus is an essential component of every living cell, biological processes influence the distribution of phosphorus in the lithosphere. Although the availability of phosphates from rocks is generally small, the local availability is influenced by three factors. The first is that rocks with higher than average phosphate content may occur over large areas, even though they form a minor constituent of the earth's crust. Secondly, phosphates may be more readily available in certain kinds of rocks than in others. According to McKelvey (1973), Hutchinson (1952) points out that phosphate is more easily liberated from sedimentary rocks than from igneous ones, because of their greater porosity and permeability. The third factor affecting

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Griffith, 1973; Sweet, 1992; Larson, 1992; McCartney, 1992). See references for more details.

phosphate availability is the environmental characteristics of the local area, such as climate, pH, and the presence of other minerals affecting the geochemistry (McKelvey, 1973). Table 2.1 contains phosphorus composition percentages for some common rock types.

Table 2.1: TP Compositions of Rock Types
(Adapted from Omernik, 1977)

ROCK TYPE		TOTAL P COMPOSITION (%)
SEDIMENTARY	Limestone	0.020
	Sandstones	0.040
	Shales	0.080
	Red Clay	0.140
	Sedimentary-mixed (average)	0.070
IGNEOUS	Rhyolite	0.055
	Granite	0.087
	Andesite	0.123
	Syenite	0.133
	Monzonite	0.139
	Diorite and Dacite	0.144
	Gabbro	0.170
	Basalt	0.244
	Igneous (averaged)	0.118

Soils

Soils are a product of their geological parent material. The concentrations and speciation of phosphorus compounds contained within soils vary widely. This is partially due to the phosphorus content in the weathered parent minerals, but is also attributed to the array of complex physical, chemical, and biological interactions within the soil-water interface.

In general, soils derived from igneous rocks have the highest phosphorus concentrations (Wetzel, 1983; Wolf, 1992). According to Bailey (1968), well drained soils also have high phosphorus levels (Wolf, 1992).

Phosphorus transport in soils primarily depends on adsorption-desorption and solubilization from the solid phase. The adsorption process is affected by factors such as: 1) chemical characteristics and organic fraction of the soil; 2) the nature of the adsorption process (i.e., physical, chemical, or both); 3) the nature of the bonds formed (i.e., Van der Waals forces, hydrophobic bonding, hydrogen bonding); and 4) the local environmental factors such as pH and temperature (Tchobanoglous and Schroeder, 1985).

Other important considerations include the cation exchange capacity (CEC) (a function of the soil type), and the presence of metals such as iron, aluminum, magnesium, and calcium (Domenico and Schwartz, 1990; Wolf, 1992). For example, clay soils have a net negative charge and high surface to volume ratio, thereby providing a high CEC. Thus metals may be readily sorbed to clay surfaces, which can facilitate complexation with phosphate compounds. Soils with high adsorptive capacities tend to become phosphorus enriched over time. The rate of phosphorus movement is a function of the degree of adsorption capacity, phosphorus-loading flux, and hydrodynamic characteristics (i.e., porosity, rainfall amounts).

Wolf's (1992) literature review concluded that the largest amounts of phosphorus carried in runoff is not from water percolating through the soil, but from phosphorus associated with sediments detached from the soil surface via erosion. According to Thompson and Troeth (1978), the upper 30 centimeters of soil in the northwestern United States contains a high percentage of total phosphorus (0.20 - 0.30 percent phosphorus as P_2O_5) in comparison to Wisconsin (0.10 - 0.19). Thus, the northwest may generally have higher phosphorus concentrations in runoff than other areas in the U.S.. Table 2.2 provides examples of total phosphorus content of some soils in Oregon in comparison with other areas.

Van Wazer (1973) pointed out that any phosphorus compound in aquatic systems can become available to biota under certain conditions, but phosphorus availability from suspended sediments can be increased thousands of times over chemical hydrolysis from enzymatic processes invoked by algae and microorganisms. Thus, eroded sediments can become a rich source of phosphorus for aquatic systems (Wolf, 1992; Van Wazer, 1973).

Table 2.2: Total Phosphorus Content of Soils from Four States
(from Wolf, 1992)

SOILS		TOTAL P (μg/g)	ORGANIC FRACTION (%)
Western Oregon Soils	Hills soils	357	65.9
	Old valley-filling soils	1,479	29.4
	Recent valley soils	848	25.6
Iowa Soils	Prairie soils	613	41.6
	Gray-brown podzolic soils	574	37.3
	Planosols	495	52.7
Arizona Soils	Surface soils	703	36.0
	Subsurface soils	125	34.0
Ohio Soils	Silty clay	715	44.9
	Silt loam	679	49.3
	Sandy loam	398	43.2

Atmospheric Inputs

Although volatile compounds involving phosphorus do not exist, contributions from wind borne particles can be significant. These particles may arise from sources such as wind blown dusts, pollens, seeds, leaves, and industrial outfall (Griffith, 1973; Salminen and Beschta, 1991). The highest atmospheric deposition of phosphorus typically occurs during the summer near industrial and agricultural areas, and lowest in remote areas during the season of highest precipitation (Wolf, 1992). Salminen and Beschta (1991) found that precipitation tends to contain higher phosphorus concentrations when storms are infrequent and of shorter duration. They also presumed that during the rainy season in western Oregon, phosphorus in precipitation is derived from mineral inputs originating from the Pacific Ocean (see Table 2.3).

Table 2.3: Atmospheric P-Loading from Forested Watersheds

Location	P-Load ($kg/km^2/yr$)	Annual Precip. (m/yr)	Reference
Average of several watersheds in Oregon	20.8	2.0	Salminen and Beschta, 1991
H. J. Andrews Exp. Forest; Western, OR.	27		Reckhow et al., 1980
Puget Sound, Wash.	22	2.0	Gilliom, 1982
Beaver Island, Mich.	21.6		Reckhow et al., 1980
Duke Forest, N. Carolina	28		Reckhow et al., 1980
Walker Branch Watershed, Tenn.	54		Reckhow et al., 1980

LAND USE VS. P-EXPORT COEFFICIENTS

Introduction

Natural steady-state background levels of phosphorus transport in drainage basins is a function of interactions among geology, soils, and climate. The topography and hydrological characteristics indigenous to the drainage basin are also major factors. Consideration of these factors would hypothetically provide enough information to empirically derive P-export coefficients for certain land use activities. This has been compiled in comprehensive studies by the EPA (Rast and Lee, 1978; Omernik, 1977; Reckhow et al., 1980) and others (Dillon and Rigler, 1975; Gilliom, 1978). For each land use activity, there is a range of P-export values. Because of the extreme complexities involved with the interactions of the previously mentioned factors, a range of P-export values is to be expected for each land use activity. In addition, poor sampling programs, procedures, and techniques can contribute to unknowns in data bias.

Nonpoint source phosphorus loads to lakes of interest should be based on export coefficients derived from watersheds with similar attributes such as climate, land use, slope, and soils. This can be accomplished by deriving export coefficients from watersheds in the same vicinity as the study area, or by selecting export coefficients from the literature which are reflective of similar attributes indigenous to the study area.

This chapter briefly addresses some of the variables involved in land use activities (i.e., forestry, agriculture, urban, residential, and septic systems) and their effects on P-export coefficients. Relevant P-export coefficients from the literature will also be listed. Particular weight will be given to the study done by Reckhow et al. (1980) because of its acceptance of coefficients only from studies which reflected good experimental design.

Forest Land Use

Species

Coniferous softwoods demonstrate higher evapotranspiration rates than hardwoods (Reckhow et al., 1980; Beschta, 1991). In their comprehensive literature review, Reckhow et al. (1980) reported that 15 years after mature deciduous hardwood watersheds in the Southern Appalachians had been converted to white pine, the annual stream flow was reduced by about 20%. Thus higher P-loads could develop from watersheds draining hardwoods compared to drainage basins containing softwoods.

Soil, Bedrock, and Parent Material

In Southern Ontario, Canada, Dillon and Kirchner (1975) reported that forested watersheds with sandy soils overlying igneous parent material had about one-half the P-export value when compared to forests with loam soils overlying sedimentary formations. Finer grained soils, such as clays and loams, have higher phosphorus adsorption capacities, and are more erodible than sands and gravel. Therefore soils and substrate combinations, such as loams and sedimentary formations could cause higher ranges of P-export coefficients (Reckhow et al., 1980).

Climate

Climate appears to play a major role in determining the export of phosphorus from forests. Areas exhibiting warm climates with high rainfall, such as the Pacific Northwest, are associated with high biological productivity. The higher amounts of precipitation contribute to increased runoff, and higher phosphorus export (Reckhow et al., 1980; Gilliom, 1981).

Drainage Basin Size

Terrestrial phosphorus loading from natural areas of a watershed is mainly from eroded soil, soil leachate, decomposed vegetative litter, and animal wastes (Gilliom, 1978). Thus, it is reasonable to expect that a lake with a very small drainage basin (with respect to lake surface area) would be dominated by near shore erosion combined with general subsurface drainage containing dissolved phosphorus. A lake with a very large drainage basin (with respect to lake surface area) would be dominated by dissolved phosphorus from general subsurface drainage because particulate phosphorus would have a greater opportunity for entrapment, and nearshore erosion would be proportionally smaller when compared to small drainage basins. Gilliom (1978) found an inverse relationship between forested drainage basin size and phosphorus loading to lakes that had a narrow range of annual runoff.

Deforestation

Vegetation and ground litter in the forest minimize surface erosion, while tree root systems bind soil masses together, contributing to soil shear strength in steep terrain. Forest vegetation uptakes free soil nutrients and provides shade, thereby minimizing stream temperature changes from solar radiation. Aquatic systems in watersheds which are altered by timber harvesting may encounter substantial increases in temperature, nutrient loading, and sedimentation.

Generally tree removal in itself has little effect on sediment concentrations in downgradient aquatic systems (Brown and Krygier, 1971). Surface erosion rates, and hence particulate phosphorus loading rates derived from forest harvesting activities, are dependent upon the methodologies used and are a function of terrain steepness (Beschta, 1978). Soil disturbances from roadbuilding, tree yarding, and slash burning contribute the most toward increases in nutrient loading and erosion. This is especially true in steep terrain where these activities can contribute to mass soil failures, thereby dramatically increasing particulate phosphorus export. Table 2.4 shows some phosphorus export coefficients from some forested watersheds which are relevant to the study area.

Table 2.4: P-Loading Coefficients from Forested Watersheds

P-Loading Coeff. (kg/km^2/yr)	Location and Investigator	Reference
85*	Siuslaw National Forest; Norris et al., 1978	Salminen and Beschta, 1991
52	average from 10 streams in the P.N.W.	Salminen and Beschta, 1991
52	H.J. Andrews Exp. Forest, Oregon; Fredrickson, 1972	Reckhow et al., 1980
68	Coyote Creek, Western Oregon; Fredrickson, 1979	Reckhow et al., 1980
18	Fox Creek, Western Oregon; Fredrickson, 1979	Reckhow et al., 1980
19.5	Using study's avg. RO in Gilliom's FORY equation	Gilliom, 1982

*This figure was found by averaging data from two studies on separate watersheds in the Siuslaw National Forest and multiplying by the average RO value for all 19 study lakes. Although it is high in comparison to general literature values, it may be relevant because the sandstone/siltstone parental material, soil, and CEC characteristics are assumed to be the same as the 19 study lakes.

Agriculture

Intensive agriculture markedly increases phosphorus export from watersheds (Dillon and Kirchner, 1975). The change from background phosphorus loading to agricultural phosphorus loading is generally proportional to the extent to which the land has been disturbed from its natural state (Prairie and Kalff, 1988). Although direct measurement is difficult (due to the diffuse nature of the pollutants), approximately two-thirds of the nations nonfederal land under cultivation, or used for grazing, contributes nearly 70 percent of the total phosphorus load (Wolf, 1992). This is primarily due to sediments, fertilizers, and animal wastes.

Agricultural phosphorus sources are difficult to assess because they are uniquely dependent on each specific situation. Factors such as soil type, fertilizer type and amounts, tillage practices, crop types, irrigation practices, grazing techniques, and animal type and density lend to difficulties in the assessment of phosphorus export coefficients. Reckhow et al. (1980) assembled extensive phosphorus export data from the literature which considers the above factors. Table 2.5 consists of data taken from Reckhow's et al. (1980) work which appear relevant to the Oregon coastal area.

Table 2.5: Agriculture/Pasture P-Loading Coefficients
(Taken from Reckhow et al., 1980)

Land Use	Location/ Soil Type	Precip. (cm/yr)	Total Phosphorus Export ($kg/km^2/yr$)	Investigator
Summer Grazed; fertilized	Ohio; silt loam	108.0	85	Chichester et al., 1979
Continuous Grazing; Some supplementary winter feeding	Maryland; well drained, sandy loam	114.7	380	Correll et al., 1977
Continuous Grazing; Active Gullies	Oklahoma; silt loams	88.3	146	Menzel et al., 1978
Continuous Grazing; Active Gullies	Oklahoma; silt loam	76.5	76	Olness et al., 1980
Agriculture & Improved Pasture	Florida; sand	96.5	110	Campbell, 1978
Agriculture, Pasture & Woodland	Ontario, Canada; silty clay ground moraine	92.5	100	Coote et al., 1978
Agriculture, Pasture & Woodland	Ontario, Canada; lacrustine clay over clay till	92.4	81	Coote et al., 1978

Urban/Residential

Urban areas contribute a wide array of pollutants generated from many different activities. Urban runoff is normally channeled into storm drains, which may carry large loads of antifreeze, oils, various particulates, pesticides and other toxic substances, fertilizers, organic trash leachate, organic litter leachate, and animal waste. The runoff constituents may stem from activities associated with industry, construction, city and residential maintenance, and atmospheric sources.

Much of the phosphorus loading from urban/residential land use can be attributed to the above activities, but is site specific as with other land uses. Characteristics such as rainfall amount, soil type, type and degree of vegetative cover, basin topography, and drainage system will affect phosphorus loading coefficients. The data in Table 2.6 appear to be representative of some of the urban conditions on the Oregon coast.

Septic Systems

Septic Systems near lakeshores are a potential major source of phosphorus loading. Effluent from septic systems typically contains about 1000 times the concentration of phosphorus in lake waters (Gilliom and Patmont, 1983). The movement of effluent phosphorus is dependent upon many factors, the most important is probably soil type, which was overviewed previously. The CEC and soil matrix are of importance because soils with low CEC and/or high permeability, such as sandy soils, allow much higher transport rates (see Soils in chapter 2 for more detail). Other important factors include: system age, seasonal ground water table relative to the drain field, distance from lake or stream, fraction of annual use, and number of people using the system. Gilliom and Patmont (1983) found that phosphorus loading to lakes was generally higher from systems which were 30-40 years old. They attributed this to: 1) inferior installation standards for older systems; 2) gradual clogging of the drainfield; 3) a deterioration of soil capacity to adsorb phosphorus between the drainfield and the lake; and 4) the long travel time for contaminated groundwater to move from the drainfield to the lake.

Table 2.6: Urban/Residential P-Loading Coefficients
(Taken from Reckhow et al., 1980)

Land Use	Location/ Soil Type	Precip. (cm/yr)	Total Phosphorus ($kg/km^2/yr$)	Investigator
Low density residential; Large lots w/Grass & Tree Cover	Michigan; Sandy loam, sandy clay loam	77.2	19	Landon, 1977
High density residential; townhouse complex; limited open space	Michigan; Sandy loam, sandy clay loam	77.2	110	Landon, 1977
High density residential cooperatives; large open grassed areas	Michigan; Sandy loam, sandy clay loam	77.2	56	Landon, 1977
Commercial, light industry and business	Michigan; Sandy loam, sandy clay loam	77.2	66	Landon, 1977
Mostly residential w/some commercial and light industry	N. Carolina; 29% impervious surfaces	108.2	123	Bryan, 1970
Mostly residential w/some commercial and light industry	Ontario, Canada;		75.7	O'Neill, 1979

CHAPTER 3: STUDY AREA

INTRODUCTION

Oregon's Coast Range extends from the Columbia River on the north, to the Klamath Mountains on the south. The southern boundary lies approximately along the Middle Fork of the Coquille River. For convenience, the Coast Range is divided into the northern and southern parts. The dividing line lies approximately along the Alsea River (Baldwin, 1981). Baldwin (1981) includes all 19 lakes of interest for this study within the northern and southern boundaries.

The general crestline altitude of the range is about 1500 feet and the summits of the passes lie east of the axis. This is due to higher rainfall on the steeper western slopes, creating more active erosion. A wavecut terrace between headlands of resistant rock and the Pacific Ocean has formed narrow coastal plains along the western edge of the Coast Range (Baldwin, 1981). All of the lakes lie on these coastal plains with the exception of Triangle and Loon Lakes, which are located just west of the Coast Range divide.

Geology

Generally, the lakes of interest and their respective drainage basins lie within areas that were derived from cenozoic marine and estuarine sedimentary rocks, and minor volcanic rocks. The primary geological parental material affecting the lakes are sandstone and siltstone; albeit some of the drainage basins may be affected by other material such as basalt, volcanic rock, and coal. Table 3.1 shows geological formations that may affect the drainage basins of interest (Baldwin, 1981).

Lake Formation

Triangle and Loon Lakes were both formed by massive landslides of Flourney sandstone and Tyee sandstone, respectively. Baldwin (1981) suggests that both lakes were formed approximately 1470 years ago during the same catastrophic earthquake.

The 17 remaining coastal lakes considered in this study were formed in association with the ocean's shoreline activity[2]. Some of the lakes were the result of bar formation across the mouths of old estuaries which were inundated by rising water levels. This phenomenon was the result of glaciation cycles causing the sea level to rise and fall (Baldwin, 1981; McGee, 1972; Johnson et al., 1985). As a result of these activities, the ancient rivers and streams are characterized by drowned mouths and valleys. Some lakes formed from this process, such as Siltcoos, Tahkenitch, and the two Tenmile Lakes, display highly dendritic features. Other coastal lakes such as Clear, Cullaby, and Cleawox were formed in a similar fashion during the glaciation cycles, or at later times, from advancing sand dune barriers (Johnson et al., 1985). Some chain lake systems were formed due to shoreline activities, in which case their physical attributes are the direct result of the topographical characteristics of the land. Mercer/Sutton, Woahink/Siltcoos, Collard/Clear/Munsel, Eel, and the Tenmile lakes represent examples of chain lake systems.

Soils

On a regional scale, Kimerling and Jackson (1985) generalize that all of the soil from the Coast Range to the ocean, north of Coos Bay, comes under the suborder of Haplumbrepts[3].This type of soil occurs in temperate to warm regions, and can be described by surface horizons darkened by high contents of organic matter, having crystalline clay minerals, with relatively high CEC under acidic conditions, and are freely drained (Kimerling and Jackson, 1985). Floras and Garrison Lake are the only exceptions, with basin soils of the suborder Haplohumults, which occur in temperate

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[2]Eckman Lake is considered a reservoir along the Alsea River. It is a water impoundment, separated from the river by Oregon Highway 34. The outflow is through a culvert.

[3]It is of interest to note that Kimerling and Jackson (1985) affix the same general soil type which occurs in the Oregon coastal region, to that of the Puget Sound region. Gilliom (1978, 1982, 1984) empirically derived phosphorus export coefficients from drainage basins in the Puget Sound region.

climates, with subsurface horizon of clay and/or weatherable minerals. They display good drainage and are mostly dark colored.

The Soil Conservation Service's General Soil Map, (1986) of the State of Oregon, delineates soil types throughout Oregon. It shows that the coastal lowlands consist of mixtures of two general soil types: 1) Bandon-Coquille-Nehalem, and 2) Templeton-Salender-Svensen. The map conveys the soils of the higher elevated forested uplands as: 1) Digger-Bohannon-Preacher, for areas south of the Alsea River, and 2) a combination of Digger-Bohannon-Preacher and Klistan-Hemcross-Harslow for Devils and Cullaby Lakes.

Larson's (1974) description of basin soils for coastal lakes in dunal regions of Lane and Douglas counties is deemed appropriate for all dunal lakes within the study area. Regional soils are principally sand or sandy loam, where pure sand dominates westerly, and to the east (between the lakes and the Coast Range), the sand gradually becomes a weakly developed sandy loam.

Climate

Mild and wet marine climatic conditions extend from the coast and into the river valleys of the Coast Range. Summer temperatures peak in August, and usually are below 70 degrees Fahrenheit. Although average temperatures range from 55-59 F, the mild winters also display raw, wet, windy and cloudy conditions. The windward slopes of the Coast Range facilitate orographic lifting. Thus precipitation generally increases with elevation, where elevations of 500-2000 feet receive the most rain. Annual precipitation ranges from 60-100 inches (1.5-2.5 m), and generally increases as one moves northerly. Winter receives the majority of total annual precipitation, as approximately 10% falls during the summer months. Wind direction generally shifts from the southwest in winter, to the northwest in summer (Kimerling and Jackson, 1985).

Forest Vegetation

Approximately 45 percent of the Oregon coastline is bordered by sand dunes

(McHugh, 1972). Through time, many of these dunes became stabilized by the procession of various kinds of vegetation. This process facilitated the development of bordering or surrounding forests along lake shorelines. The regions of higher elevation, unaffected by dunal activity and being more receptive to forest vegetation, did not require the stabilization process. Thus, two vegetation zones naturally occur along the Oregon Coast.

The Sitka Spruce Zone is confined to the coast, and has been extensively altered by logging and fire. This zone is characterized by sitka spruce, but in many places western hemlock and douglas fir dominate. Many times red alder patches form in disturbed areas and riparian situations, while western redcedar characterizes swampy habitats. Shore pine is prominent where the dunal stabilization process is occurring. The Sitka Spruce Zone naturally grades into the Western Hemlock Zone in the foothills of the Coast Range (Kimerling and Jackson, 1985).

The Western Hemlock Zones, occurring at higher elevations, are naturally characterized by mixtures of western hemlock and douglas fir, although either species may dominate. Extensive logging has occurred throughout the region, and studies have shown that vegetation communities are related to site characteristics. Other important species include the western redcedar in moist sites, and in the south, ponderosa pine and incense cedar. Where moist sites have been disturbed, red alder and bigleaf maple are common (Kimerling and Jackson, 1985).

Thermal Characteristics

The capacity and degree to which a lake thermally stratifies is generally a function of the basin morphometry and local climatic conditions. Coastal lakes that are relatively shallow, with high wind exposure, tend to be well mixed throughout the year. Examples of such lakes are: Cullaby, Devils, Eckman, Siltcoos, Tahkenitch, Floras, Garrison, and possibly Tenmile.

The remaining study lakes, which are deeper and/or more protected from wind action, become thermally stratified during the warmer months. The degree of stratification is dependent upon specific local conditions. These lakes are classified as warm monomictic, whereby complete mixing occurs after fall turnover and continues until thermal stratification begins again in early to late spring. By mid-summer the surface water becomes markedly warmer than the deeper waters, and the boundary layer (i.e., thermocline) between the two displays an abrupt thermal gradient. The warmer water above the thermocline (i.e., epilimnion) is less dense than the colder and denser water below the thermocline (i.e., hypolimnion). Thus as thermal stratification becomes more pronounced, mixing between the epilimnion and hypolimnion is reduced, and eventually is effectively stopped altogether. During fall, the lower ambient temperature and greater wind action cools the epilimnion, which increases the water's density. As the thermocline becomes less pronounced, and the epilimnion approaches temperatures similar to the hypolimnion, mixing occurs until the entire lake is "turned over", or completely mixed. Total mixing continues until spring, at which time increases in ambient temperatures and solar radiation facilitate thermal stratification again.

It is beyond the scope of this study to investigate the detailed physical, chemical, and biological interactions of lakes that thermally stratify versus lakes that remain completely mixed year round. But it should be pointed out that vast differences in physical, chemical, and biological interactions may occur between the epilimnion and hypolimnion of a thermally stratified lake, especially when the hypolimnion becomes anoxic (Wetzel, 1983). Therefore when considering the derivation of empirical phosphorus loading coefficients, the two lake types (monomictic vs. completely mixed) should be independently assessed.

Table 3.1: Profile of Oregon Coastal Lakes
(from Johnson et al., 1985; Baldwin, 1981)

	Lake	**Cullaby**	**Devils**	**Eckman**	**Triangle**	**Mercer**
	County	Clatsop	Lincoln	Lincoln	Lane	Lane
	Elevation (m)	2.3	6.1	3	211.8	9.8
	Geologic Origin/ Parental Material	*/ sedimentary; minor volcanic	*/ sandstone; siltstone; shale; basalt	**/ basalt; sedimentary; volcanic	***/ sandstone; siltstone	*/ sandstone; siltstone
Water-shed Land Use (%)	**Forestry**	93	88.9	99	92.6	90.6
	Range	1	3.4		3.5	3
	Water	4	4.3	1	0.9	5.9
	Agricult.				3	
	Urban	1	3.4		0.5	0.5
	Other	1				
	Trophic Status	eutrophic	eutrophic	eutrophic	mesotrophic	meso-eutrophic
	Comments	natural swamps & cranberry bogs	surrounded by urban; high growth pressure; upgradient cattle pastures	influx of saltwater likely	high amount of septic leaching; w/evidence of raw sewage inputs[4]	flows into Sutton; becoming culturally eutrophied

* Stream blocked by migrating sand dunes.
** Reservoir
*** Landslide

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[4] According to a representative of "Bucks Sanitary Service" (a company hired to pump sewage holding tanks), some of the sewage holding tanks adjacent to Triangle Lake, are frequently empty during the busy season. The contents of the tanks are to be treated outside of the watershed and there is evidence that some of the tanks have been emptied directly into the lake (Buchholtz, 1992).

Table 3.1 (Cont'd)
Profile of Oregon Coastal Lakes
(from Johnson et al., 1985; Baldwin, 1981)

	Lake	**Sutton**	**Collard**	**Clear**	**Munsel**	**Cleawox**
	County	Lane	Lane	Lane	Lane	Lane
	Elevation (m)	8.8	32	30.2	27.4	22.9
	Geologic Origin/ Parental Material	*/ sandstone; siltstone	*/ sandstone; siltstone	*/ sandstone; siltstone	*/ sandstone; siltstone	*/ sandstone; siltstone
Water-shed Land Use (%)	**Forestry** **Range** **Water** **Agricult.** **Urban** **Other**	89.7 3.1 6.5 0.7 	58 16 sand dunes 26%	47.9 29.2 sand dunes 22.8%	57.3 24.9 sand dunes 17.8%	47 13 sand dunes 40%
	Trophic Status	eutrophic	mesotrophic	oligotrophic	meso-oligo-trophic	oligo-trophic
	Comments	lake is two distinct basins; no T-P data on upper basin; chain lake with Mercer	affected by dunal aquifer; chain lake with Clear, Ackerley, and Munsel	affected by dunal aquifer; chain lake with Collard, Ackerley, and Munsel	affected by dunal aquifer & Ackerley, neither of which has adequate data; part of chain lakes	no apparent outflow; assumed to be affected by dunal aquifer

* Stream blocked by migrating sand dunes.
** Reservoir
*** Landslide

Table 3.1 (Cont'd)
Profile of Oregon Coastal Lakes
(from Johnson et al., 1985; Baldwin, 1981)

	Lake	Woahink	Siltcoos	Tahkenitch	Eel	N. Tenmile
	County	Lane	Lane/ Douglas	Douglas	Douglas/ Coos	Douglas/ Coos
	Elevation (m)	11.6	2.4	3.4	18.6	2.7
	Geologic Origin/ Parental Material	*/ sandstone; siltstone	*/ sandstone; siltstone	*/ sandstone; siltstone	*/ sandstone; siltstone; maybe some coal	*/ sandstone; siltstone; maybe some coal
Water-shed Land Use (%)	Forestry Range Water Agricult. Urban Other	80.9 3.1 16.6 2.5	87.9 1.3 8.7 1.1 0.5 wetlands 0.5%	88.3 2 7.3 0.1 wetlands 2.3%	89.5 10.5	93 5 2
	Trophic Status	oligotrophic	eutrophic	meso-trophic	meso-trophic	eutrophic
	Comments	chain with Siltcoos; dendritic; 15% of shoreline is in State Park	chain with Woahink; dendritic; paper mill & dam on outflow	outflow dammed by paper mill; dendritic;	dendritic; part of chain lake; no T-P data on other lake	dendritic; narrow marshes border most of lake;

* Stream blocked by migrating sand dunes.
** Reservoir
*** Landslide

Table 3.1 (Cont'd)
Profile of Oregon Coastal Lakes
(from Johnson et al., 1985; Baldwin, 1981)

	Lake	**Tenmile**	**Loon**	**Floras**	**Garrison**
	County	Coos	Douglas	Curry	Curry
	Elevation (m)	2.7	128	3	3
	Geologic Origin/ Parental Material	*/ sandstone; siltstone; maybe coal	***/ sandstone; siltstone	*/ sandstone	*/ sandstone
Water-shed Land Use (%)	**Forestry**	93	97.5	90	61
	Range				
	Water	5	0.5	5	4
	Agricult.		2	4	
	Urban	2			25
	Other			cranberry bogs 1%	sand dunes 10%
	Trophic Status	eutrophic	oligotrophic	mesotrophic	eutrophic
	Comments	chain with N. Tenmile; dendritic; bordered by narrow marshes; frequently anoxic	develops sharp thermal stratification; cabins around lake w/septic systems ambiguously delineated	well mixed; seldom stratifies;	much of drainage basin within Port Orford city limits; severe cultural eutrophication; poor watershed management

* Stream blocked by migrating sand dunes.
** Reservoir
*** Landslide

Table 3.2: Oregon Coastal Lake Data
(from Johnson et al., 1985; unless depicted)

Lake	Precip. (m/yr)	"WSA" (km^2)[1]	"A" (km^2)	"$\bar{z}$" (m)	"RO"[2] (m/yr)	"ρ" (yr^{-1})	"R"	"S"	Measured TP[3] (μg/L)
Cullaby	2.16	18	0.761	1.6	0.812	12	0.224	0.053	56.5 *
Devils	2.54	60	2.744	3	0.823	6	0.290	0.014	34 *
Eckman	2.34	15	0.182	1.2	0.349	24	0.170	0.158	55 *
Triangle	2.29	134	1.129	15.8	1.60	12	0.224	0.004	12
Mercer	2.11	22	1.453	7.1	1.41	3	0.366	0.020	21.5
Sutton	1.98	28	0.433	5.8	1.08	12	0.224	0.026	25
Collard[4]	1.98	1.2	0.14	6.7	1.80	2.3	0.399	0.281	15
Clear[4]	1.98	2.7	0.666	12.7	2.22	0.71	0.551	0.080	9.6 **
Munsel	1.91	4.4	0.445	9.3	0.941	1	0.500	0.121	14
Cleawox	1.93	4.1	0.352	5.2	1.34	3	0.366	0.115	5
Woahink	1.98	18	3.319	9.9	1.52	0.83	0.523	0.017	4
Siltcoos	2.16	169	12.81	3.3	1.50	6	0.290	0.003	39 *
Tahkenitch	2.11	83	6.775	3.3	1.62	6	0.290	0.005	17 *
Eel	1.74	25	1.437	10.5	1.21	2	0.414	0.019	6
N.Tenmile	2.13	71	4.444	3.4	1.28	6	0.290	0.008	16
Tenmile	2.12	172	6.584	3	1.38	12	0.224	0.003	13 *
Loon	2.60	221	1.19	16.3	1.05	12	0.224	0.003	4
Floras	1.78	25	0.955	5.5	1.26	6	0.290	0.023	8 *
Garrison	1.78	11.4	0.364	2.5	0.958	12	0.224	0.071	47 *

[1] Includes the watershed areas of upgradient chain lakes.
[2] Runoff is depicted as defined in equation 8.
[3] Data generally represents one sample, or two averaged during the same year. Thus it does not necessarily represent annual mean TP. Many of the samples were taken during thermal stratification.
[4] Data was taken from Christensen (1985).
* The lake generally does not develop distinct thermal stratification.
** Data represents annual mean TP (Cooper, 1985).

LAKE CHARACTERISTICS

Lake bathymetries, drainage basin characteristics, and water quality data from different studies for each individual lake of interest can be found in the appendix. Tables 3.1 and 3.2 show characteristics and data for the study lakes. Much of the information on physical characteristics was derived from Johnson et al. (1985), and is useful for gaining insight into the feasibility of individual lakes meeting model parameters. These parameters will be investigated in the following chapter. It is also of interest to observe the techniques and procedures used for data determination, as a better understanding of model uncertainties may be gained.

Lake Bathymetry

The majority of bathymetric maps (see Figures A.1 through A.19, in the appendix) were taken from Johnson et al. (1985). The dynamic processes of sedimentation can be accelerated by soil disturbances upgradient of lake basins through activities such as logging, agriculture, and construction. Unknown volumes of sedimentation could create inaccuracies in average lake depths, thus contributing to model uncertainty.

Watershed Area

Drainage basin areas were delineated by a digital planimeter on U.S. Geological Survey topographic maps (flat-map areas). They are shown in the appendix (see Figures A.1 through A.19), and were taken from Johnson et al. (1985).

Precipitation

Precipitation was determined from a statewide map prepared by the Soil Conservation Service and from the Oregon Water Resources Department drainage basin reports. Ranges of precipitation are given for large drainage basins (Johnson et al., 1985).

Flushing Rate

Flushing rate (ρ) is the reciprocal of hydraulic retention time. Johnson et al. (1985) estimated retention times (when data was available) by dividing the lake volume by the annual discharge. When discharge data was unavailable, it was estimated from U.S. Geological Survey surface runoff maps.

Annual Runoff

Annual runoff (RO) is considered to be the lake flushing rate multiplied by the lake's volume, all divided by the watershed area, or:

$$RO=\frac{\rho\,\bar{z}\cdot A}{WSA}. \tag{8}$$

These numbers are readily available from Johnson et al. (1985), except when ambiguities occurred within specific watershed hydrologies. Clear Lake is an example of ambiguous watershed hydrology caused from dunal aquifer inputs. In such cases, alternative approaches will be investigated in the following chapter.

RO may also be defined as the sum of surface and subsurface water contributing to lake inflow, shown as:

$$RO=Ppt.-ET; \tag{9}$$

where ET is evapotranspiration.

If the watershed is influenced by dunal aquifer subsurface flow, then the two previous equations may not accurately describe RO. Under such circumstances part of the hydrological regime may originate from another watershed. This is the case with the North Florence Dunal Aquifer, which affects Collard, Clear, and Munsel Lakes. The Christensen (1985) study adequately defines the flow regimes of Collard and Clear Lakes, but does not look at Munsel's hydrology.

Watershed Land Use

Land use percentages were determined from the Oregon Statewide Land Use

Inventory, which was conducted by the Oregon Water Resources Department. The Department generally used high altitude aircraft and Landsat imagery in their inventory. The following definitions are taken from Johnson et al. (1985).

Forest Land

Forest land is primarily occupied by, or used to produce trees, both deciduous and coniferous. It includes rural wood lots, regenerating cuts, and burns, as well as mixed and pure stands of merchantable or nonmerchantable timber (Johnson et al., 1985).

Range

Rangeland includes areas characterized by grasses, shrubs, meadows, unimproved pasture, and scattered trees, especially juniper or oak (Johnson et al., 1985).

Agriculture

This study combines both irrigated and non-irrigated agriculture into one group. Irrigated agriculture is land improved by artificial applications of water through flood, row, sprinkler, drip, or other irrigation techniques. Non-irrigated agriculture is land cultivated and/or harvested without benefit of irrigation (Johnson et al., 1985).

Urban

Urban land includes residential, commercial, or industrial developments, including military installations, airports, or other transportation nuclei, schools, parks, golf courses, and similar land uses (Johnson et al., 1985).

Total Phosphorus

Total phosphorus includes all chemical forms of phosphorus (dissolved and particulate, organic and inorganic) that occur in natural waters. Phosphorus was determined from a vertically integrated sample, and analyzed using a direct colorimetric (ascorbic acid) technique, as prescribed by U.S. Environmental Protection Agency procedures[5] (Johnson et al., 1985).

Chlorophyll-a

Chlorophyll-a is a characteristic algal pigment that can be used as a relative biomass indicator. The vertically integrated samples were analyzed using a fluorometric method, as prescribed by U.S.E.P.A. procedures (Johnson et al., 1985).

Water Quality Data

Selected water quality data from previous studies for the Oregon coastal lakes of interest are summarized in Table A.1 (appendix).

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[5]Johnson et al. (1985) note that nutrient and chlorophyll-a data should be used with caution because recommended holding times for these parameters were generally exceeded.

CHAPTER 4: MODEL DEVELOPMENT

INTRODUCTION

The ideal mass balance-type modeling approach empirically derives phosphorus loading coefficients for different land uses from existing data. Thus, the P-loading coefficients would represent fluxes indigenous to the study area and thereby minimize uncertainty. Gilliom (1978, 1981, 1982) was able to proceed in this manner because of the large database available for the Puget Sound, Washington region.

Reckhow et al. (1980) developed a modeling approach which facilitates lakes (and/or regions) lacking sufficient data to empirically derive P-loading coefficients. He provides P-loading tables for different land uses, taken from the literature, for his lake modeling.

Although nineteen lakes may be a sufficient number to derive realistic empirical values for land use P-loading coefficients, some lakes in this study do not meet, or are questionable in terms of meeting Vollenweider-type modeling criteria and should not be used. Therefore, a combination of the Gilliom (1978,1981,1982) and Reckhow et al. (1980) approaches appears to serve the purpose of this study best.

In this chapter, Gilliom's (1978,1981,1982) and Reckhow's et al.(1980) Vollenweider-type modeling approaches are reviewed and adapted to Oregon's coastal lakes. The study area, available data, and modeling parameters, in conjunction with both approaches, are used to optimize the predictive capabilities and minimize uncertainty.

GILLIOM METHODOLOGY

Lake Selection

The criteria Gilliom (1978) used for selecting lakes were based upon satisfying model assumptions and simplifications. The restrictions were also imposed to assure a relatively homogeneous natural environment, and to simplify nutrient loading analysis in terms of land use. Some of Gilliom's lake selection criteria are:

1. Each lake must have summer epilimnion total phosphorus data available, along with land use and physical data.

2. Less than 10% agricultural land use in the watershed and none that is riparian.

3. No major industry or commercial centers in the watershed.

4. No evidence of recent watershed clear-cutting activity.

5. Mean lake depth ≥ 2.5 meters.

6. No evidence that the lake does not completely mix during the winter period.

7. No evidence of extreme inter-basin groundwater interaction.

Approach

Gilliom's approach arranges the general model (equation 2) to solve for the total phosphorus (TP) loading term as follows:

$$L = \frac{(\bar{P})_{\infty} \bar{z} \cdot A \cdot \rho}{(1 - R)}, \tag{10}$$

and substituting equation 6 into equation 10 gives:

$$L = \frac{(\bar{P})_{\infty}}{S}. \tag{11}$$

This approach is advantageous because mean concentrations of TP are sometimes available, and are easier and less expensive to measure, than phosphorus loadings.

Most of the TP data available to Gilliom were acquired during the summer months from the epilimnion of his study lakes. Also available were data on some intensively studied lakes in his region; thus he was able to correlate mean summer epilimnion TP to annual mean TP. He found that summer epilimnion TP was about 83 percent of the annual mean TP for the lakes, and implicitly accounted for any error in a loading term (yet to be discussed). Since most TP samples were taken from the epilimnion during summer, Gilliom modified equation 10 as follows:

$$L^{*}=\frac{(\overline{P})_{ss}}{S}, \tag{12}$$

or

$$(\overline{P})_{ss}=L^{*}\cdot S, \tag{13}$$

where $(\overline{P})_{ss}$ represents the mean steady-state concentration of TP in the epilimnion of a stratified lake during the summer, in micrograms per liter, and L^{*} is the phosphorus loading rate, in kilograms per year.

Phosphorus loading rates calculated by equation 12 (L^{*}), are not equivalent to phosphorus loading rates from equation 10 (L), and could only be freely interchanged if $(\overline{P})_{ss}$ and $(\overline{P})_{\infty}$ were equal, which is generally not the case. Because the mean summer epilimnion TP was found to be approximately 17 percent less than the annual mean TP, L^{*} was expected to average 17 percent less than the actual TP loading to the lake.

When considering equation 12, one can see that all errors from the right side of the equation (sampling error and model error) are incorporated into the value of L^{*}. Gilliom statistically accounts for uncertainty through his empirically derived phosphorus loading value (L^{*}), which is further divided into background (predevelopment) sources, and cultural (human-related) sources.

The background phosphorus sources consist of water draining forested (undeveloped) areas, bulk precipitation (rainwater and dry fallout) onto the lake's surface and possible loading from the outflow of an upgradient lake. The cultural sources consist of residential runoff, seepage from septic systems, agricultural land use, and possible human related influence from upstream lakes. Gilliom defined the phosphorus loading term as:

$$L^{*}=(PREL{\cdot}A)+(FORY{\cdot}WSA_{bg})+UP+\Delta UP+\Delta RR+\Delta WW+\Delta AG; \qquad (14)$$

where

PREL= the areal rate of phosphorus loading by precipitation, in kilograms per square kilometer per year;

FORY= the phosphorus yield from forested areas, in kilograms per square kilometer per year;

WSA_{bg}= the area of land in the lake's drainage basin, where the runoff does not pass through another lake before reaching the lake of interest, in square kilometers;

UP= the background loading from an upstream lake, in kilograms per year;

and

ΔUP, ΔRR, ΔWW, and ΔAG= increases in phosphorus loading above background levels, which are respectively attributable to increased phosphorus levels in upstream lakes, residential areal runoff, nearshore septic tank systems, and agricultural land, in kilograms per year.

Background P-Loading

For lakes with no significant development in their drainage basin, equation 14 reduces to:

$$L^{*}_{bg}=(PREL{\cdot}A)+(FORY{\cdot}WSA_{bg})+UP, \qquad (15)$$

where L^{*}_{bg} is considered the loading from background or natural sources as calculated from the measured phosphorus concentration in a lake using equation 11. Gilliom evaluated background loading by considering only lakes which did not have other lakes in their drainage basin. He also determined summer precipitation loading to be approximately 20 (kg/km^2/yr). Twenty four lakes met Gilliom's background loading criteria[6], and the one remaining unknown, FORY, was calculated by rearranging equation 15 to:

$$FORY=\frac{L^{*}_{bg}-(PREL\cdot A)}{WSA_{bg}}. \tag{16}$$

Values of FORY were found to be highly correlated with annual runoff, which enabled the development of the regression equation:

$$FORY=7.1\cdot\ln(RO)+16.6 \tag{17}$$

and produced an average standard error of about 25 percent for FORY. The tools developed thus far enabled Gilliom to calculate background loading to lakes (without other lakes in the drainage basin), and to calculate the standard error, using equation 18.

$$SE_{L^{*}_{bg}}=SE_{FORY}\cdot WSA_{bg}, \tag{18}$$

where $SE_{L^{*}bg}$ is the standard error of the loading estimate. Note that all uncertainty is incorporated into the standard errors in FORY from the regression equation, and then transferred into the standard error for background loading. Gilliom's uncertainty methodology is based upon standard statistical methods as described by Meyer (1975).

Upstream Lakes

For lakes which have another lake in their drainage basin, the outlet stream from the upstream lake is considered a separate phosphorus loading source, UP. Since the upstream lake acts as a partial phosphorus trap, L^{*}_{bg} is calculated (for the upstream lake) from equation 15, and reduced by the fraction not retained in the lake.

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[6]The majority of lakes used to derive background loading had only one TP sample taken from the epilimnion.

$$UP = L^{*}_{bg} \cdot (1 - R), \tag{19}$$

where L^{*}_{bg} and R are values from the upstream lake. The standard error is approximated by:

$$SE_{UP} = SE_{L^{*}_{bg}} \cdot (1 - R), \tag{20}$$

where SE_{UP} is the standard error of loading from the upstream lake, and $SE_{L^{*}bg}$ is calculated from equation 18.

Standard error calculations for background loading from equation 15 can be assessed by:

$$SE_{L^{*}_{bg}} = \sqrt{(SE_{FORY} \cdot WSA_{bg})^2 + SE^2_{UP}}. \tag{21}$$

Cultural P-Loading

Cultural phosphorus loading values can be calculated by evaluating the differences between background loading and present day loading.

$$L^{*} - L^{*}_{bg} = \Delta UP + \Delta RR + \Delta WW + \Delta AG. \tag{22}$$

Present day loading (L^{*}) is calculated from equation 12, and L^{*}_{bg} was already discussed. Gilliom found that when 4 or more phosphorus concentration samples were available, then standard error could be calculated by standard statistical methods. Otherwise, standard error was estimated from the following equation (Gilliom, 1978):

$$SE_{(\overline{P})_{ss}} = \left(\frac{0.30}{\sqrt{n}} + 0.20\right) \cdot (\overline{P})_{ss} \tag{23}$$

where n is the number of samples available. A reasonable standard error of lake sensitivity, S, was   20 percent. Thus, the standard error of L^{*} can be calculated as:

$$SE_{L^*}=\sqrt{\frac{SE^2_{(\bar{P})_{ss}}}{S^2}+\frac{(\bar{P})^2_{ss}\cdot SE^2_S}{S^4}} \tag{24}$$

Gilliom evaluated cultural phosphorus loading by first calculating ΔUP and its standard error using the same procedures described previously. Then he progressively isolated and considered lakes with ΔRR, ΔWW, and ΔAG inputs respectively, while uncertainties were calculated in a cumulative fashion as discussed earlier. His database of useable lakes was sufficiently large to enable him to develop empirical relationships for ΔRR and ΔWW, which reasonably compared to the literature values. He also found that the magnitude of phosphorus loading is correlated to septic tank age, which is evident in the literature.

RECKHOW METHODOLOGY

Model Approach

The steady state solution that Reckhow et al. (1980) uses for the mass balance on a lake's phosphorus concentration is fundamentally equivalent to Gilliom's approach, but differs with respect to the phosphorus retention coefficient. Reckhow based his model on the assumption of a constant phosphorus settling velocity such that the phosphorus mass balance (equation 1) would be expressed as:

$$V\cdot\frac{dP}{dt}=M-v_s\cdot P\cdot A-Q\cdot P. \tag{25}$$

Where: P = average annual total phosphorus concentration in lake (mg/L)

M = annual mass rate of phosphorus inflow to lake (kg/yr);

v_s= apparent phosphorus settling velocity (m/yr);

V = lake volume ($10^6 m^3$)

A = lake surface (bottom) area (km^2)

Q = annual inflow to lake ($10^6 m^3/yr$)

Thus, the rate of phosphorus deposited to the sediments is a function of the bottom (surface) area. The steady state model becomes:

$$P=\frac{L_R}{v_s+\bar{z}\cdot\rho}, \tag{26}$$

where[7]: L_R= M/A = phosphorus surface loading (g/m^2/yr);

$\bar{z}\cdot\rho$= q_s = surface overflow rate (m/yr).

Using linear regression on his data set, Reckhow's model resulted in the following form:

$$P=\frac{L_R}{11.6+1.2q_s}, \tag{27}$$

where the empirically derived model error (s_{mlog}) for the log transformed model is 0.128.

NOTE: The units in some of Reckhow's variables are different than those used by Gilliom. They have not been changed for this investigation because accuracy may be lost when converting the empirically derived coefficients in equation 27, and the procedure for finding model error was based on the logarithmic tranformation of terms. For a comprehensive description of this modeling approach, Reckhow (1979, 1980, 1983) is recommended.

Model Criteria

Reckhow et al.'s (1980) model was developed from 47 northern temperate lakes included in the EPA's National Eutrophication Survey. He recommends that the model be applied to lakes possessing characteristics within the same range as those lakes used to develop the model. His lakes displayed characteristics within the following boundary conditions:

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[7] L_R denotes the loading term in the Reckhow approach, which has different units than Gilliom's loading term.

1. lakes lie within the northern temperate zone.
2. 0.004 mg/L $\leq$ P $\leq$ 0.135 mg/L;
3. 0.07 g/m^2/yr $\leq L_R \leq$ 31.4 g/m^2/yr;
4. 0.75 m/yr $\leq q_s \leq$ 187 m/yr.

Furthermore, he suggests that caution be used when applying his model to lakes with special characteristics, such as:

5. shallow lakes (less than approximately 3 meters);
6. closed lakes (no apparent outflow);
7. lakes with heavy aquatic weed growth.

Lakes outside the range of model criteria and/or displaying special characteristics may create uncertainties of unknown magnitude. Thus, caution should be used if such lakes are addressed.

Modeling/Uncertainty Procedures

This particular model (Reckhow, (1979) labeled it Quasi-General) was derived from a wide range of lake types, enabling its application to a broad scope of lakes. Furthermore, some lakes which do not meet all model criteria, such as shallow or closed lakes, may still be facilitated, although to a lower or unquantified degree of uncertainty. The analysis procedure estimates variables in the following order: Step 1) surface overflow rate (q_s); Step 2) areal phosphorus loading (L_R); Step 3) lake phosphorus concentration (P); Step 4) phosphorus prediction uncertainty (s_T).

<u>Step 1:</u> Estimation of q_s

The areal water loading (q_s) is estimated by:

$$q_s = \frac{Q}{A} = \bar{z} \cdot \rho; \qquad (28)$$

where

and; Ppt. = mean annual precipitation (m/yr).

$$Q=(WSA\cdot RO)+(A\cdot Ppt.); \tag{29}$$

All of the above lake variables are given, or can be solved for explicitly from data in Johnson et al. (1985).

Step 2: Estimation of L_R

Reckhow et al. (1980) compiled a survey of phosphorus export coefficients from different land uses that were screened according to acceptable criteria and are representative of good sampling design. His procedure recommends the selection of high, most likely, and low export coefficients for each land use. The high and low loading estimates represent additional phosphorus loading error to be added to model error, enabling total uncertainty to be calculated.

Caution should be used when selecting high and low loadings because much of the error in the loading estimates is already incorporated into the model error (Reckhow et al., 1980). Thus, a poor choice of export values decreases the accuracy of uncertainty calculations. To maximize uncertainty accuracy, Reckhow provides warnings of possible bias for export coefficients. He stresses that when appropriate, the warnings should be addressed by increasing the high and/or low loading estimates. The criteria provided for export coefficient selection are the descriptive conditions, some of which are given in Tables 2.5 and 2.6, and consist of details such as: land use activity, vegetation type, soil/surface characteristics, location, precipitation, annual runoff, and fertilization rate. Extrapolation of high and low export coefficients should reflect the modeler's confidence in correlations between the application lake watershed characteristics and those given in the literature. For example, when the modeler knows that the "most likely" export coefficient chosen was determined from a good sampling with similar watershed characteristics, in comparison to the lake of interest, then the high and low values should be selected to reflect little uncertainty. Furthermore, Reckhow suggests that a single "most likely" precipitation loading coefficient is adequate unless the precipitation loading approaches approximately 25%

of the total loading. In essence, his methodology for the selection of different land use export coefficients is a somewhat subjective process which relies on similarities between watershed characteristics from the literature and application watersheds. It also is based on the modeler's knowledge, experience, and/or professional intuition.

The rate of phosphorus flowing to a lake is estimated in the same fashion as the Gilliom approach (equation 14). Each nonpoint source's loading coefficient is multiplied by its respective area and then summed, as follows:

$$\begin{gathered} M=(FORY \cdot WSA_{bg})+(PREL \cdot A)+(E_{C_{ag}} \cdot WSA_{ag}) \\ +(E_{C_{u}} \cdot WSA_{u}) \\ +(E_{C_{st}} \cdot (no.capita\text{-}years) \cdot (1-S.R.)) \\ +P.S.I.; \end{gathered} \tag{30}$$

where: FORY, PREL, Ec_{ag}, and Ec_U = export coefficients for: forest land, atmospheric, agricultural land, and urban area respectively, ($kg/km^2/yr$)

Ec_{st} = Export coefficient to septic tank systems impacting the lake, (kg/capita-yr/yr)

WSA_{ag} = Area of agricultural land (km^2)

WSA_U = Area of urban land (km^2)

# of capita-years = # of capita-years in the watershed, of septic systems which impact the lake

S.R. = soil retention coefficient (dimensionless)

PSI = point source input (i.e., industrial, sewage treatment plant, etc.), (kg/yr)

Ec_{st} differs from the other export coefficients in that it represents the estimated annual amount of phosphorus transported to the septic system, not the lake.

The soil retention coefficient (S.R.) estimates the effectiveness of the soil in immobilizing phosphorus between the septic system and the lake. This coefficient ranges from 0 to 1.0. Zero represents soil in which all phosphorus eventually reaches the lake or stream, and conversely 1.0 indicates that all phosphorus is immobilized by the soil.

Reckhow suggests four major aspects of watershed soils that affect phosphorus immobilizing capabilities and influence contact duration time that should be considered in S.R. selection. These are phosphorus adsorption capacity, natural drainage[8], permeability, and slope. In addition, four general mechanisms attributed to phosphate removal in the soil column are rapid removal or adsorption, slow mineralization and insolubilization, plant uptake, and biological immobilization. Reckhow et al. (1980) found that formation of insoluble iron and aluminum compounds and adsorption of phosphate onto clay are the most important phosphorus immobilization mechanisms. He suggests that a single ("most likely") S.R. coefficient is sufficient if the estimated loading from septic systems is less than approximately 25% of the total phosphorus load. Otherwise, additional "low" and "high" S.R. coefficients are necessary.

Estimation of number of capita-years is based on the distance from the water body that septic systems may impact phosphorus loading. Conditions that determine the size and location of the impact zone include soil type, drainage patterns, water tables, and slopes. Population surveys or projections are useful when assessing the lake's current or future status, respectively. Reckhow recommends that high and low loading estimates should be based on the uncertainty of population projections. The total number of capita-years is calculated by summing the permanent and seasonal resident capita-years, as follows:

$$\begin{aligned} \textit{Tot. \# capita-yrs} = &(\textit{Permanent \# capita-yrs}) \\ &+(\textit{Seasonal \# capita-yrs}); \end{aligned} \tag{31}$$

where,

$$\text{Permanent capita-years} = \text{avg. \# people per living unit} \cdot \text{\# days at unit per year} \cdot \text{\# of living units}$$

[8]Natural drainage is related to the water table depth. An aeration zone between the drainfield and the water table is required to effectively immobilize phosphorus. The greater the aeration zone depth, the greater the likelihood of phosphorus immobilization.

The seasonal capita-years are calculated in the same fashion as the permanent capita-years, but with different projection coefficients for the first two terms on the right side of the equation.

Reckhow notes that shallow lakes possessing anoxic bottom waters may contribute appreciable amounts of phosphorus from the sediment/water interface to the overlying waters. In this case low, most likely, and high loading estimates should be used to estimate additional phosphorus inputs.[9]

Once the low, most likely, and high export coefficients have been established for each loading source, equation 30 is used to calculate the respective annual mass of phosphorus inflow to the lake. The low, most likely (ml), and high phosphorus surface loading rates (L_R) are found by:

$$L_{R_{(low),(ml),(high)}} = \frac{M_{(low),(ml),(high)}}{A}. \tag{32}$$

Step 3: Calculation of P

The lake phosphorus concentration may now be calculated by substituting the values of q_s and $L_{R(high,\, ml,\, low)}$ into equation 27 as follows:

$$P_{(high),(ml),(low)} = \frac{L_{R_{(high),(ml),(low)}}}{11.6+1.2 \cdot q_s} \tag{33}$$

Step 4: Uncertainty Predictions

The uncertainty estimation approach is based on first order error analysis.

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[9]Reckhow's approach does not provide for phosphorus loading from upstream chain lakes. This can easily be rectified by adding the percentage of phosphorus not retained in the upstream lake as outlined in Gilliom's methodology.

Uncertainty for all terms in the model (L_R, v_s, q_s), and in the model itself are needed for complete evaluation. In most applications Reckhow found the uncertainty in v_s to be small. Moreover, since q_s is a function of hydrologic variability and flow measurement error, which also affect L_R, the uncertainty in both model variables generally tend to cancel each other because they are in the denominator and numerator, respectively. Exceptions occur in lakes with highly variable, or poorly characterized, flushing rates. His uncertainty analysis makes provisions for all cases and will subsequently be investigated.

The uncertainty analysis procedure is based on the following assumptions:

1. The model error is initially expressed in log-transformed concentration units, but may be combined with the variable error terms once the transformation is removed.

2. The range ("high" - "low") for phosphorus loading error is approximately two times the standard deviation, where about 90% of the distribution is contained within 2 standard deviations.

3. The individual error components are described adequately by their standard deviations (variances).

Step 4a: Calculation of log $P_{(ml)}$

The logarithm of "most likely" phosphorus concentration is taken, $P_{(ml)}$.

Step 4b: Estimation of "positive" model error, (s_m^+)

The model error (s_{mlog}) has been determined to be 0.128. The "positive" model error (s_m^+) is calculated by:

$$s_m^+ = antilog(\log P_{(ml)} + s_{mlog}) - P_{(ml)}. \tag{34}$$

Step 4c: Estimation of "negative" model error, (s_m^-)

Step 4d: Estimation of "positive" loading error, (s_L^+)

$$s_m^- = antilog\cdot(\log P_{(ml)} - s_{mlog}) - P_{(ml)} \tag{35}$$

The loading error estimate must be converted into compatible units with the model error. The "positive" loading error (s_L^+) is calculated by:

$$s_L^+ = \frac{P_{(high)} - P_{(ml)}}{2}. \tag{36}$$

<u>Step 4e:</u> Estimation of "negative" loading error, (s_L^-)

$$s_L^- = \frac{P_{(ml)} - P_{(low)}}{2}. \tag{37}$$

<u>Step 4f:</u> Estimation of Areal Water loading error,(s_{qs}^+, s_{qs}^-)

This step is used only if the lake of interest has a highly variable flushing rate, or if the watershed hydrology is poorly characterized (i.e., extreme interbasin groundwater interaction). It is assumed that the uncertainty originates in the flow term (Q), and is re-expressed as q_s=Q/A. Uncertainty in the prediction of total phosphorus concentration due to uncertainty in q_s is:

$$s_{q_s} = \sqrt{\frac{1.44 \cdot L_R^2}{(11.6+1.2 \cdot q_s)^4} \cdot s^2(q_s) - \frac{2.4 \cdot L_R}{(11.6+1.2 \cdot q_s)^3} \cdot s(q_s) \cdot s(L_R)^{\pm} \cdot \delta(L_R, q_s)} \tag{38}$$

where:

1. s_{qs} = contribution to the total phosphorus concentration prediction uncertainty due to uncertainty in q_s. This term has positive and negative components as follows:

   a. s_{qs}^+ is found by using $\delta(L_R,q_s)$, $s(q_s)$, and $s(L_R)^+$ in equation 38.

   b. s_{qs}^- is found by using $\delta(L_R,q_s)$, $s(q_s)$, and $s(L_R)^-$ in equation 38.

2. $\delta(L_R,q_s)$ is the correlation between L_R and q_s, which is primarily determined by Q. Therefore, the correlation should be positive, and diminish the importance of the q_s uncertainty contribution. Reckhow concluded from cross-sectional studies that the correlation coefficient between L_R and q_s ranges from +0.5 to +0.8.

3. $s(q_s)$ is the uncertainty estimate for q_s, as determined by the analyst. (Note that it is different from s_{qs}).

4. $s(L_R)$ is the uncertainty estimate for L_R. It has positive and negative components which can be expressed by using the high, most likely, and low phosphorus loading terms calculated in step 2:

$$s(L_R)^+ = \frac{L_{R\,(high)} - L_{R\,(ml)}}{2}; \tag{39}$$

and

$$s(L_R)^- = \frac{L_{R\,(ml)} - L_{R\,(low)}}{2}. \tag{40}$$

Step 4g: Estimation of total "positive" uncertainty, (s_T^+)

$$s_T^+ = \sqrt{(s_m^+)^2 + (s_{L_R^+})^2 + (s_{q_s}^+)^2}. \tag{41}$$

Step 4h: Estimation of total "negative" uncertainty, (s_T^-)

$$s_T^- = \sqrt{(s_m^-)^2 + (s_{L_R^-})^2 + (s_{q_s}^-)^2}. \tag{42}$$

Step 4i: Confidence Limits

The uncertainty analysis approach used by Reckhow et al. (1980) allows the confidence limits to be written as:

$$\textit{Prob.}\ [(P_{(ml)} - s_T^-) \leq P \leq (P_{(ml)} + s_T^+)] \geq 0.55. \tag{43}$$

Equation 43 states that about 55% of the time, the actual average total phosphorus concentration lies within the bounds defined by the prediction plus or minus the prediction uncertainty. This interpretation can be broadened to the 90% confidence limit range by:

$$\textit{Prob.}\ [(P_{(ml)} - 2 \cdot s_T^-) \leq P \leq (P_{(ml)} + 2 \cdot s_T^+)] \geq 0.90, \tag{44}$$

where equation 44 states that approximately 90% of the time, the actual average total phosphorus concentration is bounded by the prediction plus or minus the prediction uncertainty.

COASTAL LAKE MODELING PROCEDURE

Introduction

Vollenweider-type models are most accurate and have lowest uncertainty when land use phosphorus loading coefficients are quantitatively derived directly from the region of interest. Uncertainty in predictive capabilities can be bound through linear regression, which is a function of the database. Complex phosphorus dynamics coupled with heterogeneity (or incomplete knowledge) of P-transport media require a large database of lakes within the region.

Gilliom (1978,1981,1982) studied a total of 52 lakes in the Puget Sound region. 24 of the lakes met criteria to mathematically describe background phosphorus loading (FORY). The prediction of background (pre-development) conditions is essential if the modeler is interested in the theoretical natural trophic state of a lake. This state may be used by lake managers as a base from which land use plans are derived.

The approach proposed by Reckhow et al. (1980), as previously described, provides different land use phosphorus loading coefficients from the literature, enabling the modeler to make TP predictions. This procedure also provides the means to calculate uncertainty, thereby giving lake managers a basis for land use decisions. The phosphorus loading coefficients and uncertainty calculations are based on a somewhat subjective selection process, and may be less accurate than Gilliom's approach.

Approach

In the Oregon Coastal Lake Study (OCLS), the database is not sufficient to accurately estimate background phosphorus loading. Although it is assumed that 17 of the 19 lakes of interest[10] lie within a region of similar conditions (i.e., climate, geologic parental material, soil, vegetation and, topography), many of the lakes either do not, or are questionable for meeting modeling criteria (described earlier in the chapter), and/or the database for each lake does not adequately represent the lake for modeling purposes. Qualifying characteristics of the OCLS lakes are:

1. Cullaby Lake: The depth and extensive macrophytes are beyond modeling

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[10]Triangle and Loon Lakes are near the summit of the Coast Range. This contrasts the other lakes which are generally located in the coastal lowlands.

criteria so that the lake should not be considered further.

2. Devils Lake: The extensive macrophytes require that the lake not be used for P-loading calibration.

3. Eckman Lake: The lake is too shallow, therefore it is unusable in the OCLS.

4. Triangle Lake: Characteristics of location and geological formation, with respect to the other lakes, may cause P-loading coefficients to be significantly different. Outside Reckhow's "q_s" modeling criteria, therefore should not be considered further.

5. Mercer Lake: Significant residential and septic system input; cannot be used for background P-loading derivations.

6. Sutton Lake: The lake consists of two separate basins. For modeling purposes, each basin should be considered as separate lakes. TP data are only available for the downgradient basin, therefore Sutton Lake cannot be included in this study.

7. Collard Lake: Dunal aquifer, residential and septic system inputs require that caution be used for background predictions.

8. Clear Lake: Development in specific watershed is minimal but must account for Collard input. Dunal aquifer influences on watershed hydrology requires the use of caution for background predictions.

9. Munsel Lake: Significant residential and septic system input; most downgradient lake on Collard/Clear/Ackerley chain; no known TP data on Ackerley exists; also influenced by dunal aquifer; cannot be used for background P-loading derivations.

10. Cleawox Lake: Residential and septic system inputs exist, along with heavy recreational use during summer months.[11] No apparent surface outflow exists, therefore it should not be used for P-loading derivations, and any model predictions should be viewed with caution.

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[11]Both Cleawox and Woahink Lakes are partially within Honeyman State Park. The park provides numerous campsites with shower and bathroom facilities, which significantly affect P-loading in the area. Information on quantity of sewage, and the location of the septic system(s) was unavailable from the Lane County Sanitation Dept..

11. Woahink Lake: Residential and septic system inputs exist, also heavy recreational use during summer months. Should not be used for background predictions.

12. Siltcoos/Tahkenitch/Tenmiles: Same as Devils Lake.

13. Eel Lake: A lake which is downgradient in a chain-lake system. TP inflow from the upgradient lake (Clear Lake, Douglas Co.) are required, but are not available. Therefore Eel Lake cannot be modeled.

14. Loon Lake: Different proximity to the ocean, and geological formation than other lakes may produce different P-loading coefficients (Triangle and Loon are near the summit of the Coast Range). Outside Reckhow's "q_s" modeling criteria, therefore should not be considered further.

15. Floras Lake: The presence of cranberry bogs and macrophytes requires that the lake not be used for P-loading calibration.

16. Garrison Lake: The depth, extensive macrophytes, and Reckhow's "L_R" are outside modeling parameters. Therefore the lake will not be considered further.

Lack of TP sampling data, combined with a small pool of acceptable lakes which meet model criteria, required that the best modeling approach for this study was to incorporate both Gilliom's, and Reckhow's methodologies. This was accomplished by employing (when possible) Gilliom's method to derive "most likely" phosphorus loading coefficients for background and cultural loading. Each coefficient was compared to literature values while considering conditions indigenous to the study area. This approach maximized accuracy, and minimized subjectivity. Once "most likely" loading coefficients were established, Reckhow's method was followed to provide uncertainty calculations. Until more data are available for the lakes of interest, the initial modeling requires the majority of the "most likely" loading coefficients to be taken directly from the literature.

Calibration of "most likely" P-loading coefficients was applied only to the lakes adequately meeting Vollenweider-type modeling criteria. These lakes are: Mercer, Collard, Clear, Munsel, and Woahink. Once acceptable P-loading coefficients were established, they were applied to the remaining lakes, which in the worst case scenario are questionable or borderline in meeting modeling criteria. These lakes are Cleawox, Devils, Siltcoos, Tahkenitch, the Tenmiles, and Floras; their uncertainty calculations may not be accurately quantified.

All of the lakes which meet modeling criteria do not develop distinct thermal stratification. Therefore it was considered most appropriate to use P-loading coefficients which are based on an annual mean TP concentration. This approach facilitates the model's use for all acceptable lakes.

Background P-Loading

Clear Lake is the only lake which is minimally influenced by cultural activities. Therefore it is the only lake acceptable for the derivation of a "most likely" FORY loading coefficient. Collard Lake must also be considered because of its upgradient influence on Clear Lake. Background phosphorus loading may be found by including dunal influence in equation 15 and rearranging as follows:

$$FORY=\frac{L_{bg}-(PREL\cdot A)-(DUNL\cdot Area_{s.d.})-UP}{WSA_{bg}}, \tag{45}$$

where DUNL = phosphorus loading from dunal aquifer, (kg/km^2/yr);

$Area_{s.d.}$= sand dune area, (km^2).

Since data are minimal for establishing PREL and DUNL, phosphorus loading coefficients must be subjectively chosen from literature values.

<u>Precipitation (PREL)</u>

Available data on precipitation phosphorus concentrations within the Oregon coastal region are very limited. Some observations throughout the years have been reported by the National Atmospheric Deposition Program (NADP), but virtually all samples were below the phosphorus detection limits. Since the lakes are affected by the same weather patterns (caused from proximity to the ocean), coupled with similar land use activities, it is assumed that PREL is constant throughout the study region.

The most relevant data on annual atmospheric TP deposition was the average concentration taken from four forested watersheds in Western Oregon (Salminen and Beschta, 1991). The average PREL was about 21 kg/km^2/yr with an average annual precipitation of 2.0 m/yr. This PREL compares well with literature values such as in the Gilliom study (Puget Sound, Washington area), where the annual mean phosphorus loading was 22 kg/km^2/yr. The value (21 kg/km^2/yr) is also reasonable because the mean annual precipitation (2.0 m/yr) compares well with the study area. The mean annual precipitation for all 19 lakes is approximately 2.1 m/yr (Johnson et al., 1985). When considering all factors involved it appears that 21 kg/km^2/yr is the "most likely" PREL for the OCLS.

"High" and "low" estimates of PREL are not required because 25% of the annual TP load cannot be attributed to atmospheric inputs within the study area.

Dunal (DUNL)

Christensen (1985) measured TP in the Clear Lake dunal area and found the concentration to be 0.007 mg/L, which converts to 13.3 kg/km^2/yr (with an annual precipitation of 1.9 m/yr). Although the above concentration appears to be taken from only a single sample, the validity seems representative of DUNL based on the following logic.

The accepted TP guidelines for oligotrophic status is generally less than 0.010 mg/L (Dillon and Rigler, 1975; Reckhow et al., 1980; Gilliom, 1978, 1981,1982). Conversion of this phosphorus concentration (0.010 mg/L) into a loading term (DUNL) requires RO. RO as described by equation 9 requires an ET value. Christensen (1985) concluded that an acceptable ET value for a lake's surface and upland areas, in the Clear Lake region to be 2 ft/yr. Thus, a reasonable RO, via equation 9, is 1.3 m/yr. Combining the general oligotrophic boundary phosphorus concentration with RO produces a loading term of 13.2 kg/km^2/yr, which is virtually the same as Christensen's sample value.

It appears logical to assume that phosphorus loading from dunal areas is minimal, as it represents natural background loading from a highly permeable medium with a low CEC. Alternatively, the presence of high iron content in soils of some coastal regions is known, which could greatly affect phosphorus dynamics. The only dunal phosphorus loading data found were from Christensen's (1985) work, therefore, until more data are available, 13.3 kg/km^2/yr appears to be the "most likely" DUNL.

It is also assumed that "high" and "low" values are not needed for uncertainty calculations because the range for other loading terms should more than compensate for any uncertainty unaccounted for in $DUNL_{(ml)}$.

Forests (FORY)

The Christensen (1985) study described the complex hydrological regime (including the North Florence Dunal Aquifer) in the Collard/Clear Lake area. This information provided the most accurate derivation of phosphorus retention coefficients (R), lake sensitivity coefficients (S), and flushing rates (ρ), which are critical in FORY calculations. Results from Christensen's report also provided background annual mean TP for Clear Lake ranging from 5 to 6 micrograms per liter. If the mean of this range (5.5 μg/L) is divided by the sensitivity coefficient (S) (equation 12), then a background loading coefficient (L_{bg}) may be established. It should be noted that in this case the background TP represents an annual mean concentration. Thus, the FORY derived would reflect TP on an annual basis, rather than from the epilimnion of a thermally stratified lake, and be applicable to all lakes in the study.

Collard Lake influences on Clear Lake are represented as:

$$UP=(1-R)\cdot[(FORY\cdot WSA_{bg})+(PREL\cdot A)+(DUNL\cdot AREA_{sd})]. \qquad (46)$$

Using the Christensen (1985) hydrological profile and the other substitutions, equation 45 gives a FORY of about 40 kg/km^2/yr for Collard and Clear Lake. This compares reasonably with FORY literature values taken from similar vegetation and watershed characteristics. Table 2.4 contains some of these values.

When considering unknown uncertainty factors involved in the Clear/Collard Lake FORY calculation, it is appropriate to use a wide range of FORY coefficients.

This would not only reflect uncertainty, but could be implicitly used to provide insight into TP changes caused by logging activities. Where high FORY values could represent recent clear cuts and generally poor forestry practices, and low FORY values may represent forests in their natural state. It is apparent that the coastal region generally yields a higher FORY than other regions, probably due to sandier soils, sedimentary parental material, climate, among other factors. Therefore "high", "most likely", and "low" FORY coefficients are assumed to be 75, 40, and 25 kg/km^2/yr, respectively.

Cultural P-Loading

Residential Loading (RESL)

Gilliom (1982) was able to empirically derive a P-loading coefficient for residential runoff, but it appears low (7.0 kg/km^2/yr) when compared to the literature. From the tables provided in Reckhow et al. (1980), the most applicable value for residential runoff was 19 kg/km^2/yr (Landon, 1977). While Nelson (1990) used a value of 30 kg/km^2/yr for residential runoff in the Collard Lake watershed.

Lack of data and/or study lakes with acceptable characteristics for empirically deriving a residential loading coefficient, necessitates that a value be assumed from the literature. General conditions, as previously described (soils, geological parental material, RO, etc.), appear to be more conducive to phosphorus transport than in the Gilliom study. Thus, until more information is available, Nelson's value (30 kg/km^2/yr) appears to be representative of a "most likely" loading coefficient for residential areas. This figure is assumed to represent a semi-rural setting, where trees, grass, and general vegetative cover exist on large low density lots. It is also assumed to account for a minor part of the overall P-loading to the lakes (less than 25%), so "high" and "low" constituents will not be required.

It should be noted that P-loading from ΔRR represents loading in addition to natural (background) sources, which is written as:

$$\Delta RR = RESL \cdot WSA_{res} \tag{47}$$

where: RESL = 30 kg/km^2/yr

WSA_{res}= residential area in watershed (as described above)[12]

Nearshore Septic Systems (SEPL)

Collard, Munsel, and Cleawox were the only study lakes in the Johnson et al. (1985) atlas where cultural P-loading was minimized to residential and septic systems. Munsel and Cleawox cannot be considered for empirical derivations because: 1) Adequate data are unavailable for Ackerley Lake, which is a chain lake upgradient to Munsel Lake, and 2) Cleawox's hydrological profile is considered inadequate because it has no apparent outflow (Johnson et al., 1985). Using Collard Lake in conjunction with Gilliom's method, to derive a septic system P-loading coefficient (SEPL), gives results that do not make sense. Therefore, a literature investigation is required for establishing SEPL, and all remaining P-loading coefficients.

Until more information is available correlating septic systems, their age, and P-loading coefficients, it appears most logical to designate a general value to each dwelling unit. The Nelson (1990) and Christensen (1985) studies used a septic system P-loading coefficient (SEPL) of 0.8 kg/Dwelling Unit/yr for the Clear/Collard Lake area. This figure was taken from Gilliom's work, and approximately represents the amount of TP reaching the lake from a 30 to 40 year old septic system in the Puget Sound region. Gilliom assumed that only septic systems within 75 meters (250 feet) of the shoreline affected TP concentrations within the lake. Furthermore, he assumed that each system accommodated an average of 2.5 people per year, which is consistent with the Florence area (Christensen and Rosenthal, 1982; Nelson, 1990) and that each

[12] This study has found that accurate accounts of WSA_{res} and the number of nearshore septic systems may be difficult to assess, being a function of each county's record keeping system.

person contributes approximately 1.5 kg/yr of T-P to the septic system (Gilliom, 1978; Reckhow et al., 1980)

When considering factors such as soil, CEC, and water table elevation with respect to drainfields for the Oregon coastal region versus Puget Sound, a higher P-loading coefficient would be expected. Thus, it appears that a "most likely" SEPL of 1.0 kg/D.U./yr would be a reasonable and conservative estimate, and that values of 1.3 and 0.9 would be representative of "high" and "low" coefficients (if needed for projections).

It should be noted that the amount of TP loading from septic systems is generally expected to be a function of time (Gilliom, 1982; Christensen, 1985). But for this study, a time dependent SEPL coefficient is unfeasible because: 1) in some counties, there is no record of older septic systems (greater than 15 to 20 years old), and 2) in some cases, the needed information is dispersed between State, County, and local agencies. This makes septic system data (proximity to shoreline, and age) extremely difficult to obtain, with questionable results because of item 1. Thus it appears that septic system input is best represented for this as:

$$\Delta WW = SEPL \cdot (D.U.) \tag{48}$$

where: SEPL = 1.0 kg/D.U./yr

D.U. = Number of dwelling units within 75 meters (250 ft.) of the shoreline

Range/Agriculture (RANGL)

Johnson et al. (1985) delineate between rangeland, irrigated, and non-irrigated agricultural land uses. Although contributing minimally overall to P-loading, the major land use under this category (for the study area) is rangeland, with a minor part attributed to non-irrigated agriculture. Reckhow et al. (1980) provide a table consisting of P-loading coefficients from "mixed agricultural"watersheds, which seems to be the most fitting for this general type of land use. Thus, when considering the combination of pasture, hay, woodlands, and crops, a P-loading coefficient (RANGL) of 100

$kg/km^2/yr$ appears generally appropriate. Hence, a mass per time basis, range/agriculture P-loading inputs may be represented as:

$$\Delta AG = RANGL \cdot WSA_{ag} \tag{49}$$

where: RANGL = 100 $kg/km^2/yr$

WSA_{ag} = rangeland and agricultural land in the watershed (km^2)

Since the overall phosphorus contribution is minor for this type of land use, "high" and "low" coefficients are not required.

Urban (URBL)

P-loading from urban inputs (URBL), as with rangeland and agriculture land uses, can be highly variable and unique to the factors and practices of that particular area. The Atlas of Oregon Lakes (Johnson et al., 1985) description of urban land use is fairly general (see chapter 3), and does not include information about sewage treatment or percentage of impervious surfaces. This study assumes P-loading coefficients representing urban areas delineated in the "Atlas" are: 1) facilitated with sewage treatment; 2) not heavily industrialized; and 3) a minor portion of the land is covered with impervious surfaces. When considering all factors, a "most likely" URBL appears to be 90 kg/km /yr, with a range of 130 to 70 for "high" and "low" coefficients respectively. The latter two values would be required only if urban inputs were projected to be greater than 25% of the total P-loading. P-loading inputs from urban areas is represented as:

$$\Delta URBAN = URBL \cdot WSA_{urb} \tag{50}$$

where: URBL = 90 $kg/km^2/yr$

WSA_{urb}= urban area in watershed (km^2)

Wetlands/Bogs (WETL)

Wetlands and bogs represent a minor part of the land usage in several of the study lakes. Although they appear to be insignificant with respect to total P-loading from their respective watersheds, a relatively high coefficient seems most representative in both cases. This assumption is based on the descriptions given in Johnson et al. (1985). Some wetlands (Siltcoos and Tahkenitch) are described as marshes which contain a rich variety of natural vegetation and support great numbers of wildlife and birds[13]. Total phosphorus from the animal and bird excretion would contribute to a higher P-loading coefficient. In the case of cranberry bogs (Cullaby and Floras), it is assumed that fertilization occurs. Thus, a high P-loading coefficient would be expected. Since both land uses contribute a minor part of the total P-loading in their respective watersheds, they were combined for this study.

P-loading information was not found for either type of land use in the literature, therefore the relatively high value of 150 kg/km^2/yr was assigned to WETL. P-loading from wetlands or bogs are represented as:

$$\Delta BOG = WETL \cdot WSA_{bog} \tag{51}$$

where: WETL = 150 kg/km^2/yr

WSA_{bog}= wetland or bog area in watershed (km^2)

Table 4.1 shows the nonpoint source P-loading coefficients which are considered applicable to this study. Since the values generally represent TP concentrations from lakes on an annual mean basis (rather than during thermal stratification), they may be applied to all the study lakes. Again, it should be emphasized that uncertainty cannot be accurately quantified for lakes which do not meet the modeling criteria discussed previously.

[13] It should be noted that phosphorus-loading due to waterfowl can be significant, but was not assessed for this study.

Table 4.1: OCLS P-Loading Coefficients

P-loading Coeff.	"High" ($kg/km^2/yr$)	"m.l." ($kg/km^2/yr$)	"low" ($kg/km^2/yr$)
FORY	75	40	25
PREL		21	
DUNL		13	
RESL		30	
URBL	130	90	70
RANGL		100	
WETL		150	
SEPL	0.9 kg/DU/yr	1.0 kg/DU/yr	1.3 kg/DU/yr

OCLS Modeling/Uncertainty Procedure

Using the P-loading coefficients from Table 4.1 in conjunction with land use areas and lake characteristics enables phosphorus predictions and uncertainty calculations to be made. Available data and circumstances (as previously described) necessitates the use of Reckhow's Quasi-General modeling methodology, which is a four step process. The steps are summarized below and include some minor requirements needed to facilitate the OCLS.

Step 1: Calculate q_s (eq. 28)

$$q_s=\bar{z}\cdot\rho \tag{28}$$

Step 2a: Calculate phosphorus loading rate (eq. 30)

$$M_{(high),(m.l.),(low)} = (FORY \cdot WSA_{bg}) + (PREL \cdot A) + (DUNL \cdot AREA_{s.d.}) + UP + \Delta UP + \Delta WW + \Delta RR + \Delta AG + \Delta URBAN + \Delta BOG; \tag{52}$$

where "low" and "high" constituents of a P-loading coefficient is required if the particular land use contributes greater than 25% of total P-loading.

Step 2b: Calculate surface loading rates (eq. 32)

The OCLS surface loading rate may be found by using equation 32 with a conversion factor, so that the units will correspond with Reckhow's model.

$$L_{R_{(low),(m.l.),(high)}} = \frac{[M_{(low),(m.l.),(high)}] \cdot 0.001}{A}; \tag{53}$$

where the units are:

L_R = g/m^2/yr

M = kg/yr

A = km^2

Step 3: Calculate P (eq. 33)

$$P_{(high),(m.l.),(low)} = \frac{L_{R_{(high),(m.l.),(low)}}}{11.6 + 1.2 \cdot q_s} \tag{33}$$

Step 4: Uncertainty calculations

Uncertainty calculations for the OCLS are as prescribed in steps 4a through 4i, in Reckhow's modeling/uncertainty section of this chapter.

CHAPTER 5: MODEL RESULTS AND DISCUSSION

INTRODUCTION

Johnson et al. (1985) is the only source which provides land use delineation and TP for the lakes of interest during the same time period (the early 1980's). Since updated land use delineation within each watershed may require extensive field work, it is considered outside the bounds of this study. For the purposes of the OCLS, unless otherwise noted, land use delineation is assumed to be unchanged from the information given in Johnson et al. (1985). This data are given in table 5.1.

Table 5.1: Watershed Land Use Delineation
(from Johnson et al., 1985)

Lake	WSA_{bg} (km^2)	WSA_{res}[1] (km^2)	WSA_{ag} (km^2)	WSA_{urb} (km^2)	WSA_{bog} (km^2)	WSA_{sd} (km^2)	D.U.[2] (1980)	D.U. (92)
Devils	53.3	?	2.04	2.04	0	0	?	?
Mercer	19.9	0.27	0.66	0.11	0	0	86	92
Collard	0.788	0.105	0	0	0	0.272	18	21
Clear	0.525	0	0	0	0	0.309	0	0
Munsel	1.21	0.081	0	0	0	0.045	14	16
Cleawox	1.93	0.46	0	0	0	1.82	13	15
Woahink	14.6	0.35	0	0.45	0	0	50	?
Siltcoos	134	0.065	4.06	0.395	0.845	0	40	40
Tahkenitch	73.3	0.005	1.66	0.083	1.91	0	3	3
N. Tenmile	66	?	0	1.42	0	0	?	?
Tenmile	93.9	?	0	2.02	0	0	?	?
Floras	22.5	?	1.0	0	2.21	0	?	?

[1] WSA_{res} is a rough approximation. Information was not obtained for lakes outside of Lane and Douglas Counties.

[2] D.U. in both columns signifies nearshore dwelling units with septic systems. In Douglas County, D.U. information was only available for present circumstances and was assumed to be unchanged from 1980.

Johnson et al. (1985) did not provide nearshore septic system and WSA_{res} information. A search at Lane and Douglas County offices provided data on these subjects to varying degrees. The accuracy was dependent upon each county's record keeping procedures. For example, in Lane County delineation between 1980 and present nearshore septic systems was possible, but only a rough approximation of WSA_{res} was found. In Douglas County nearshore septic systems were approximated by assuming that nearshore lots with buildings assessed over $10,000 had septic systems. WSA_{res} was roughly approximated by assuming that only nearshore lots contributed to ΔRR, and that each lot was 17,500 ft^2 (see Table 5.1).

Using the information from Tables 4.1, 5.1, and 5.2 in conjunction with the modeling procedure enables phosphorus concentration predictions and uncertainties to be made for each lake of interest. The predictions are representative of water quality conditions concurrent with the time that watershed land usage was delineated (early 1980's). Fortunately, these predictions may be compared with TP sampling data gathered at the same approximate time (Johnson et al., 1985)(see Table 5.3 and Figure 5.1). If new development within the lake's watershed is minimal between 1980 and the present (which appears to be the case for nearshore dwelling units in Lane County), then TP predictions may also reflect present conditions.

Table 5.2: Constant Lake Characteristics
(from Johnson et al., 1985)

Lake	Mean Depth, $\bar{z}$ (m)	Flushing Rate, ρ (yr^{-1})	q_s (m/yr)	Retention (R)	Sensitivity (S)
Devils	3	6	18.0	.290	.014
Mercer	7.1	3	21.3	.366	.020
Collard[1]	6.7	2.3	15.5	.399	.281
Clear[1]	12.7	0.71	9.02	.551	.080
Munsel	9.3	1	9.3	.500	.121
Cleawox	5.2	3	15.6	.366	.115
Woahink	9.9	.83	8.22	.523	.017
Siltcoos	3.3	6	19.8	.290	.003
Tahkenitch	3.3	6	19.8	.290	.005
N. Tenmile	3.4	6	20.4	.290	.008
Tenmile	3	12	36	.224	.003
Floras	5.5	6	33	.290	.023

[1] The data for Collard and Clear Lake were taken or derived from the Christensen (1985) study.

OCLS MODEL RESULTS

Table 5.3 displays the annual mean TP predictions from both the Gilliom and OCLS methods. The low and high boundaries are the 55% and 90% confidence limits for annual mean TP respectively, (see Chapter 3) are applicable to the OCLS method only. Gilliom's method is unqualifiable, but is of interest for comparative reasons.

All but four lake's (Clear, Siltcoos, Cleawox, and Floras) measured TP are within the 55% confidence limits. If the P-loading estimates are within 50% of the measured value, then the prediction is considered to have a relatively high reliability

(Gilliom, 1982). Thus, modeling predictions of the eight lakes within the confidence limits are considered highly reliable. Clear and Siltcoos had measured TP concentrations that fell within the 90% confidence limits, which is moderately reliable (Gilliom, 1982). Cleawox and Floras displayed low predictive reliability.

Figure 5.1 shows Gilliom's and OCLS's predicted versus measured TP datapoints for the lakes of interest. They may be compared to the "predicted = measured" line, which theoretically represents optimal conditions for TP sampling, modeling parameters, and lake characteristics. Best fit lines[14] (linear regression) depicting OCLS's and Gilliom's respective TP predictions versus measured TP are also shown in Figure 5.1. In both cases the r^2 is 0.73.

The figure shows measured TP plotted along the X-axis and the calculated (predicted) TP plotted on with respect to the Y-axis. Thus, horizontal distances between the data points and the "predicted = measured" line may be representative of annual mean TP measuring inaccuracies. While the vertical distance between the data points and the "predicted = measured" line may represent model prediction inaccuracies, it is unknown whether the TP inaccuracies, or the model prediction inaccuracies carry more weight, and will remain so until accurate annual mean TP data are available. In the cases of Floras and Cleawox, it appears that most inaccuracies can be attributed to the measured TP, because each was a single TP sample of unusually low magnitude.

Five of the twelve lakes are represented by only one TP sample, and six of the lakes are represented by averaging two TP samples taken from Johnson et al. (1985) Clear Lake's TP concentration was the annual mean, taken from Cooper (1985). With the exception of Floras and Cleawox, it appears that the measured TP, as shown in Table 5.3, is reasonably representative of the annual mean TP concentrations during that time.

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[14]Cleawox and Floras were not included in "Best Fit" calculations because they were outside the 90% confidence limits.

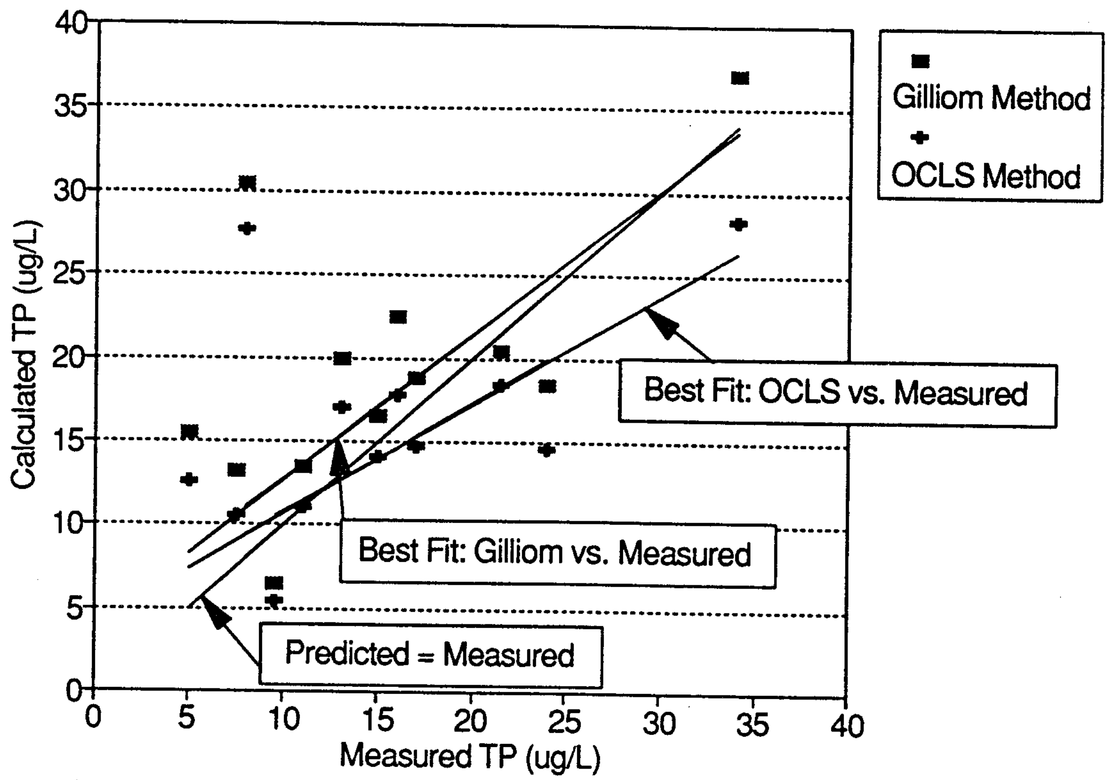

Figure 5.1: Predicted TP vs. Measured TP

Table 5.3: OCLS Predictions
(based on data from the early 1980's)

Lake	Meas. TP[1] (μg/L)	Gilliom TP (μg/L)	OCLS TP (μg/L)	55% Low Conf. Limit	55% High Conf. Limit	90% Low Conf. Limit	90% High Conf. Limit	Atlas Trophic Status[3]	Predicted Trophic Status[4]
Devils	34	37.1	28.3	19.8	42.4	11.4	56.5	E	E
Mercer	21.5	20.4	18.5	13.0	27.5	7.5	36.5	M-E	M-E
Collard	15	16.5	14.1	10.2	19.9	6.4	25.7	M	M
Clear	9.6[2]	6.5	5.5	4.0	7.7	2.5	9.9	O	O
Munsel	11	13.5	11.0	7.9	15.7	4.9	20.5	O-M	O-M
Cleawox	5	15.4	12.5	9.0	17.8	5.6	23.2	O	O-M
Woahink	7.5	13.2	10.6	7.5	15.7	4.4	20.8	O	O-M
Silt.	24	18.5	14.6	10.2	22.0	5.8	29.4	E	M-E
Tahken.	17	18.7	14.8	10.4	22.1	5.9	29.5	M	M
N. Ten.	16	22.4	17.8	12.3	27.3	6.8	36.7	E	M-E
Tenmile	13	20.0	16.9	11.7	25.9	6.5	34.8	E	M-E
Floras	8	30.5	27.6	19.8	40.1	11.9	52.5	M	E

[1] Measured TP may not be reflective of annual mean TP. TP from Devils, Collard, Cleawox, N. Tenmile, and Floras represent one sample. The remaining lakes' TP represent the average of a sample taken in spring and one taken during fall.

[2] The measured TP from Clear Lake represents annual mean TP, taken from Cooper (1985).

[3] Trophic classification is based on a trophic state index adapted from Carlson (1977). Please see Johnson et al. (1985) for more information.

[4] O = oligotrophic (TP ≤ 10 μg/L); M = mesotrophic (10 μg/L ≤ TP ≤ 20 μg/L); E = eutrophic (TP ≥ 20 μg/L). Adapted from Dillon and Rigler (1975) and Reckhow et al. (1980).

OCLS Trophic Status Predictions

The trophic status predictions provided in Table 5.3 are reflective of the range between the low and high boundaries for the 55% confidence limits. Using Reckhow's et al. (1980) classification system (see footnote to Table 5.3), the predicted trophic status is generally the same as described by Johnson et al. (1985) (Atlas). The greatest difference was Floras where eutrophic status was predicted, while the Atlas classified it as mesotrophic.

Other differences were from Siltcoos and the Tenmile Lakes. The three were predicted to range between mesotrophic and eutrophic, while the Atlas shows all three as eutrophic. This may be attributable to the extensive macrophytes disrupting normal phosphorus sedimentation and/or to the highly dendritic characteristics of the lakes.

Cleawox and Woahink were predicted to range between oligotrophic and mesotrophic, while the Atlas considered both oligotrophic.

With the exception of Floras, the trophic status prediction appears to generally be representative of the actual conditions during the early 1980, as described in the Atlas.

Background Predictions

Background predictions were calculated for each lake by rearranging equation 52. The background P-loading was represented by forest, precipitation, dunal, and upgragient lake inputs as follows:

$$\begin{aligned} M_{back(low)(ml)(high)} &= M_{(low)(ml)(high)} \\ &-[\Delta UP+\Delta WW+\Delta RR+\Delta AG+\Delta URBAN+\Delta BOG] \end{aligned} \tag{54}$$

The procedure was continued, as prescribed in Chapter 4, and the results are displayed in Table 5.4.

In five of the twelve lakes the most likely background phosphorus predictions are higher than the measured concentrations. Again, it is unknown what this differentiation is attributable to, but it appears that the range between the confidence limits is generally reliable. This conclusion is based upon the previous discussion, where comparisons were possible between predictions and measured TP concentrations. Unfortunately, that is not possible in this case because background concentrations are theoretical, but the data in Table 5.4 may be helpful in assisting land use decisions by providing a base for theoretical background conditions.

Table 5.4: OCLS Background Predictions

Lake	Measured TP (µg/L)	Predicted Backgrd. TP (µg/L)	55% Low Conf. Limit	55% High Conf. Limit	90% Low Conf. Limit	90% High Conf. Limit
Devils	34	24.0	16.5	37.2	8.9	50.3
Mercer	21.5	15.3	10.5	23.6	5.7	31.9
Collard	15	9.0	6.3	13.5	3.6	18.1
Clear	9.6	4.6	3.3	6.6	2.1	8.6
Munsel	11	8.8	6.3	13.0	3.7	17.1
Cleawox	5	10.2	7.3	14.9	4.3	19.6
Woahink	7.5	9.2	6.4	13.9	3.6	18.7
Silt.	24	13.4	9.3	20.5	5.1	27.7
Tahken.	17	14.0	9.8	21.2	5.5	28.4
N. Ten.	16	17.1	11.7	26.3	6.4	35.6
Tenmile	13	16.2	11.1	25.0	6.1	33.8
Floras	8	25.6	18.2	37.5	10.8	49.4

Mercer Lake Case Study

The OCLS model may be used to assist in land use planning decisions by providing the means to predict TP under different land use scenarios. Several scenarios are provided in Table 5.5, using Mercer Lake as an example.

Referring to Table 5.5, measured TP is the average of two TP concentrations from Johnson et al. (1985) and is 14% higher than the predicted value based on early 1980's land use conditions. Mercer Lakes's predicted background TP concentration (calculated without cultural P-loading inputs) is about 25% less than the predicted TP for current conditions. Measured TP for current conditions (average of two samples; Dagget, 1992) and the prediction for current conditions (assuming no forest clear cut and 92 nearshore septic systems) are within 5%. The scenario for sewering Mercer lake's watershed (excluding P-loading from septic systems) was found to decrease TP by about 9% from the predicted current conditions. The addition of 50 nearshore resident lots and septic systems to current conditions increased TP by about 5%. Clear cutting 25% of the forest increased TP concentration by approximately 17% above the predicted TP for current conditions (based on the "high" P-loading value of 75 $kg/km^2/yr$).

Based on OCLS model predictions, the current TP of Mercer Lake is 20% above the background TP, and if the watershed could be returned to background conditions the water quality would be mesotrophic (moderate biological productivity), never reaching oligotrophic status (biologically unproductive). The increased TP caused from an additional 50 nearshore septic systems may appear small in comparison to current conditions, but illustrates the effect of incremental development in the watershed. It should be noted that the predictions are based on longer term effects and may not become apparent rapidly. Considering increases in P-loading as a function of time (see chapter 4; Nearshore Septic Systems), and that the septic system P-loading coefficient (SEPL) is based on properly installed and functioning septic systems, TP may eventually be higher than predictions show. The greatest water quality degradation due to forested clear cuts is generally realized over a relatively short term and decreases over time as soils stabilize. The Mercer Lake clear cut scenario represents TP conditions which may occur in the short term. It appears that the "high" P-loading coefficient used (75 $kg/km^2/yr$) is reasonable for clear cutting, but could substantially increase as a function of soil disturbances and/or factors such as roadbuilding, poor harvesting techniques, steep terrain, and non-reforestation.

Table 5.5: Mercer Lake P-Loading Scenarios (OCLS TP Predictions)

SCENARIO	TP (most likely) (μg/L)	55% Low Confidence Limit	55% High Confidence Limit
Measured (Atlas, 1981) (average)	21.5		
Predicted Early 1980's Conditions	18.5	13.0	27.5
Predicted Background Conditions	15.3	10.5	23.6
Measured Current Conditions (Dagget, 1992)	20.0		
Predicted Current Conditions[1]	19.2	13.6	28.4
Current Conditions w/Sewering Around Lake[2]	17.5	12.2	26.3
Add 50 Nearshore Sep. Sys.[2] (above current cond.)	20.2	14.3	29.6
25% of Forest Land is Clearcut[3] (above current cond.)	22.4	15.2	31.5

[1] Assumes no clear cut areas and that 92 nearshore septic systems exist compared to 86 D.U. from 1981. Other land use percentages remain the same as the early 1980's.

[2] Does not consider nutrient loading caused from construction, road building, etc.. Lots are considered to be 17,500 ft^2 on average.

[3] The "m.l." P-loading coefficient for clear cut areas was assumed to be 75 $kg/km^2/yr$.

CHAPTER 6: CONCLUSIONS & RECOMMENDATIONS

SUMMARY AND CONCLUSIONS

A Vollenweider-type modeling approach similar to Gilliom's (1979,1981,1982) may be the most desirable for a collection of lakes within the same geographical area. Gilliom's model provides predictive capabilities and uncertainty calculations based upon data taken from the study area. Although the lakes from this study are generally considered to be within the same geographical area, with similar geological parental materials, soils, and climate, the database was not sufficient to adequately derive P-loading coefficients for different land uses. Therefore an approach similar to Gilliom's has been adapted for this study.

The model developed for the OCLS used data from the study area to derive (when possible), or give insight toward the selection of valid land use P-loading coefficients. TP prediction and uncertainty calculations were based on the work of Reckhow et al. (1980).

Of the nineteen original lakes considered for the OCLS, Cullaby, Eckman, Triangle, Sutton, Eel, Loon, and Garrison were excluded. These seven lakes either displayed characteristics beyond the boundary conditions for Vollenweider-type modeling, or insufficient data were available to adequately model the lake.

The OCLS model for eight of the remaining twelve lakes (all but Clear, Siltcoos, Cleawox, and Floras) has relatively high predictive reliability, while the predictive capabilities for Clear and Siltcoos are moderately reliable (Gilliom, 1982). The OCLS model should be used with caution for Cleawox[15] and Floras because they displayed low prediction reliability.

These reliabilities are partially based on modeling procedures, and the accuracy

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[15]The low predictive reliability for Cleawox may be a function of the lack of data on the hydrological profile within the watershed. Although it is considered a "closed" lake, adequate information on the subsurface flow is unavailable.

of: 1) land use P-loading coefficient selection for the study area, and 2) land use delineation (as described in Chapters 4 and 5). Furthermore, the relative reliability of the OCLS model is also based upon the sampling techniques and accuracy of the annual mean TP concentration of the lakes taken during watershed land use delineation. Calibration of the OCLS model was partially based on a comparison of measured annual mean TP with predicted annual mean TP. Thus, improper TP sampling procedures or inaccurate annual mean TP concentrations could give biased information to an unknown degree.

Water quality data from all known previous studies on the nineteen Oregon coastal lakes were assembled and can be found in Tables A.1 through A.19 in the appendix. Other relevant information, such as physical characteristics of the lakes and watershed, and P-loading data can be found in the tables throughout this report.

The modeling approach and calibration was based upon adequacy of water quality data and lakes which met modeling criteria (see Chapter 4). The lakes which were acceptable for consideration were Devils, Mercer, Collard, Clear, Munsel, Cleawox, Woahink, Siltcoos, Tahkenitch, N. Tenmile, Tenmile, and Floras. Much of the water quality data (phosphate, nitrate, chlorophyll-a, and secci-depth) in the appendix were not directly useable for the modeling approach selected for the OCLS. Vollenweider-type lake mass balance modeling is the only acceptable modeling approach which could be used with the existing database, in conjunction with the requirement that water quality predictions be qualified with uncertainty calculations. It is also based upon annual mean TP concentrations[16] and therefore is the most relevant water quality parameter. With the exception of Clear Lake (Cooper, 1985), the TP data for the acceptable lakes generally does not validly represent annual mean concentration. Furthermore, the Atlas of Oregon Lakes (Johnson et al., 1985) was the only study found which furnished TP data and delineated percentages of watershed land uses. Thus model calibration was partially based on annual mean TP predictions

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[16] Lakes that thermally stratify may use a mass balance based upon the mean TP concentration of the epilimnion during stratification (Gilliom, 1982).

(summing all individual land uses, multiplied with their respective P-loading coefficients, for each watershed) versus measured annual mean TP. The "best fit" linear regression line for the model's predicted TP versus measured TP had an r^2 of 0.73. The predictive capabilities of the model were generally considered highly reliable because the TP predictions were usually within 50% of the measured value (Gilliom,1982).

This study found that forest land use activities can significantly degrade the water quality of a lake. This is due to the greater percentage of forest land use within each watershed, and can be related to the model by increasing the P-loading coefficients (using $FORY_{\text{"high"}}$) when soil disturbances or other forest management related activities occur. Other activities may potentially create larger decreases in water quality (such as construction, urbanization, agriculture, recreation, etc.) because of their greater respective P-loading magnitudes, but this does not generally appear to be the case within the study region.

CONCLUSIONS

Based on the results of this study, the following conclusions can be drawn:

1. The existing water quality database for Oregon coastal lakes is sporadic and insufficient to accurately characterize present water quality conditions or to assess water quality changes that may have occurred due to watershed land use changes.

2. Watershed land use data for the selected Oregon coastal lakes reflects early 1980s conditions and need to be updated to better estimate current and future phosphorus loading rates.

3. Phosphorus loading coefficients for forestry, dunal aquifer, and precipitation were derived from data specific to the Oregon coastal region. Other phosphorus loading coefficients were estimated based on literature values related to Oregon coastal conditions.

4. The OCLS phosphorus mass-balance model, calibrated from site specific data and literature values, can estimate water quality trends for Oregon coastal lakes but has a relatively large uncertainty.

5. The OCLS model shows promise as a tool to assist in land use management decisions by estimating water quality effects of projected land use changes.

6. A case study of Mercer Lake, used to illustrate the water quality effects of selected land use changes in the Mercer Lake watershed, predicted that a 9% reduction in lake TP could be achieved from sewering the lake, while a 5% and 17% increase in lake TP would result from adding 50 nearshore dwelling units and clear cutting 25% of the forested watershed, respectively.

7. Because an average of 81% of Oregon coastal lake's watersheds are in forest land use, forest management activities may have the greatest overall impact on water quality. However, the greatest relative water quality changes occur due to urban development (including septic systems for waste disposal) and agricultural activities.

RECOMMENDATIONS FOR FUTURE WORK

Johnson et al. (1985) was the main data source for concurrent coastal lake TP concentrations and land use delineations. Lake TP concentrations were generally based on one, or two samples averaged, and may not represent annual mean TP. Although it appears that the OCLS model is relatively reliable, accuracy can be increased and uncertainty can be reduced. Therefore it is recommended that:

1. A water quality program be initiated to monitor chemical, biological, and physical characteristics of Oregon coastal lakes (e.g., TP, chlorophyll-a, macrophytes, algae and zooplankton, temperature and dissolved oxygen profiles) on a regular basis (e.g., bi-monthly from November to April, semi-monthly from May to October).

2. A program be initiated which periodically updates land use changes within Oregon coastal lake watersheds. This information would be used in conjunction with water quality data to more accurately calibrate the OCLS model and enhance model predictability.

3. A study be initiated to derive site specific P-loading coefficients for Oregon coastal lake watersheds. Such a program would serve to more accurately calibrate the OCLS model and reduce uncertainty in its predictions.

4. A study be established to describe the subsurface hydrology of Oregon coastal lake dunal aquifers. This is especially needed for the Florence Dunal Aquifer (Collard, Clear, Ackerley, Munsel), and the Cleawox Lake watershed.

5. For water quality studies involving chain lakes, that all lakes in the chain be included in the sampling program.

6. A study be initiated which correlates mean summer epilimnetic TP to annual mean TP. Once the correlation is established, the sampling program could be reduced to those lakes which thermally stratify, and TP samples would only be required during thermal stratification.

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APPENDIX

Table A.1 contains water quality data from previous known studies for all lakes of interest.

Figures A.1 through A.19 contain drainage basin and bathymetric maps of the lakes of interest. All maps were taken from Johnson et al. (1985). The dot on each bathymetric map represents the sample site used by the Atlas. The bathymetric maps which have numbers represent the location of sample sites which are different from the Atlas sites, and correspond to studies as referenced in Table A.1.

Table A.1: Water Quality Data From Previous Studies

Lake	Sample Date	Source	Sample Site	Sample Depth (m)	TP (mg/L)	Phosphate PO_4-P (mg/L)	Nitrate NO_3-N (mg/L)	Chlor.-a (μg/L)	Secci Depth (m)	Comments
CULLABY	6/22/92	Dagget, 1992	Atlas	JBS[1]	0.06			8.75	0.69	DEQ samples
	7/27/92	Dagget, 1992	Atlas	JBS	0.06			13.2	0.91	DEQ samples
	7/31/89	Sweet, 1990	Atlas	JBS	.099	.007	.04	13.5	1.0	
	10/6/89	Sweet, 1990	Atlas	JBS	.096	.011	.08	11.8	1.5	
	4/17/82	Johnson et al., 1985	Atlas	JBS	.038			2.2	0.9	DEQ samples
	10/16/82	Johnson et al., 1985	Atlas	JBS	.075			3.1		DEQ samples
	6/3/69	McHugh, 1972	Atlas	JBS		.007	.09			DEQ samples
	8/19/69	" "	"	"		.04	.16			" "
	10/17/69	" "	"	"		.02	.19			" "
ECKMAN	8/17/82	Johnson et al., 1985	Atlas	JBS	.055			6.5	1.5	" "
TRIANGLE	5/13/82	" "	Atlas	JBS	.012			2.1	3.5	" "
	8/12/70	McHugh, 1972	Atlas	"		<0.003	0.06			" "
	7/7/70	Powers et al., 1975	?	4 meters	0.017	.007	<0.01	4.7	2.2	
	8/5/70	" "	?	" "		<0.002	<0.01	3.6	3.5	
	9/2/70	" "	?	" "	0.009	0.003	<0.01	3.5	2.5	
		Maloney et al., 1975	?	?		0.04				
	6/12/70	Smith and Bella, 1973	?	3 meters	0.015	0.003	0.022			

[1] JBS = just below surface

Table A.1 Cont'd: Water Quality Data From Previous Studies

Lake	Sample Date	Source	Sample Site	Sample Depth (m)	TP (mg/L)	Phosphate PO_4-P (mg/L)	Nitrate NO_3-N (mg/L)	Chlor.-a (µg/L)	Secci Depth (m)	Comments
DEVILS	1990-1991	CH2M HILL	Atlas	0.5	0.060		0.08	41.4		mean values
	1990-1991	" "	"	5.0	0.100		0.10	43.4		" "
	1986	Thomas et al., 1990	Atlas	0.5	0.023	0.014	0.164			" "
	1987	" "	"	5.0	0.022	0.13	0.167			" "
	1986	" "	"	0.5	0.041	0.019	0.162			" "
	1987	" "	"	5.0	0.023	0.013	0.170			" "
	7/17/81	Johnson et al., 1985	"	JBS	0.034			2.5	2.1	DEQ samples
	1981	KCM, 1983	"		0.04	0.01	0.23	4.9	2.0	annual mean
	1971	Kavanagh, 1973	"	0.5	0.025					mean value
SUTTON	7/19/81	Johnson et al., 1985	"	JBS	0.025			8.8	2.8	DEQ samples
	11/19/82	" "	"	"	0.032			5.8	1.7	" "
	9/23/80	DEQ	?	?		0.009	0.26			sampling by DEQ
	Sept. 1978	Bryant et al., 1979	Atlas	3		0.030	0.02	12.1	1	
	Feb. 1979	" "	"	1		0.028	0.8	2.6	2.2	
	June 1979	" "	"	1		0.040	0.84	2.2	3.8	
	Aug. 1979	" "	"	1		0.094	<0.1	6.6	3	
	4/3-6/73	Larson, 1974	"	1-2	0.03	0.06	0.71		2.7	sampling by DEQ
	6/11-12/73	" "	"	"	0.13	0.08	0.27		3.0	" "
	8/27-28/73	" "	"	"	0.03	0.01	0.01		2.7	" "
	8/11/60	Oakley, 1962d	?						2.7	from Larson, 1974
	7/13/60	Saltzman, 1961d	?						3.0	" "
	1/23/48	" "	?						3.4	" "

Table A.1 Cont'd: Water Quality Data From Previous Studies

Lake	Sample Date	Source	Sample Site	Sample Depth (m)	TP (mg/L)	Phosphate PO_4-P (mg/L)	Nitrate NO_3-N (mg/L)	Chlor.-a (µg/L)	Secci Depth (m)	Comments
MERCER	6/23/92	Dagget, 1992	Atlas	1	0.02			5.33	4.06	DEQ Lab
	7/29/92	" "	"	1	0.02			4.34	4.88	"
	7/6/89	Sweet, 1990	"	1	0.025	0.006	0.51	4.1	5.1	
	9/24/89	" "	"	1	0.044	<0.01	0.07	15.5	3.4	
	10/7/89	Citizen Watch	"	1					3.9	from Sweet, 1990
	9/24/89	" "							4.1	" "
	8/25/89	" "							3.8	" "
	7/18/81	Johnson et al., 1985	"	JBS	0.030			9.2	2.5	DEQ Lab
	5/2/82	" "	"	JBS	0.013			9.9	2.1	" "
	7/21-22/80	DEQ				0.015	0.58			
	Aug. 1979	Bryant et al., 1979	Average of sta. 2, 3, 4	1		0.053				
	June 1979	" "	" "			0.067				
	Feb. 1979	" "	" "			0.022				
	Sept. 1978	" "	" "			0.057				
	3/20/72	Larson, 1974	Atlas	1-2		0.19	0.87			DEQ Lab
	6/13/72	" "	"	"		<0.01	0.38		4.0	" "
	8/21/72	" "	"	"		<0.01	<0.03		4.9	" "
	10/30/72	" "	"	"		0.02	0.17			" "
	2/13-15/71	Kavanagh, 1973	"	0.5	0.110					
	3/20-22/71	" "	"	"	0.0063					
	6/9-12/71	" "	"	"	0.0042					
	6/26-29/71	" "	"	"	0.0060					

Table A.1 Cont'd: Water Quality Data From Previous Studies

Lake	Sample Date	Source	Sample Site	Sample Depth (m)	TP (mg/L)	Phosphate PO_4-P (mg/L)	Nitrate NO_3-N (mg/L)	Chlor.-a (µg/L)	Secci Depth (m)	Comments
Mercer cont'd	8/5/68	Larson, unpublished							4.6	from Larson, 1974
	8/15/60	Oakley, 1962							4.0	" "
	7/13/60	Saltzman, 1961							2.7	" "
	1/16/48	" "							3.0	" "
COLLARD	8/16/82	Johnson et al., 1985	Atlas	JBS	0.015			4.2	3.0	DEQ Lab
	Sept. 1978	Bryant et al., 1979	"	1		0.003	0.02	2.7	4	
	Feb. 1979	" "	"	1		0.012	0.02	1.2	6	
	June 1979	" "	"	1		0.097	0.5	1.3	5	
	Aug. 1979	" "	"	1		0.091	<0.1	1.6	5.5	
	6/12-13/72	Larson, 1974	sta. 2	1-2		<0.01	0.25		4.6	DEQ Lab
	8/21-22/72	" "	" "	"		0.02	<0.03		5.5	" "
	10/30-31	" "	" "	"		<0.01	0.16		3.4	" "
	8/10/60	Oakley, 1962							6.1	from Larson, 1974
CLEAR	1984	Cooper, 1985	average	JBS	0.0088		0.05			mean epilimnion during stratificatio n
	1984	" "	average	JBS	0.0096		0.11			annual mean
	5/12/82	Johnson et al., 1985	Atlas	JBS	0.010			2.0	3.5	DEQ Lab
	11/19/82	" "	"	"	0.016			2.4	5.0	" "
	Sept. 1978	Bryant et al., 1979	"	1	0.010		0.03	1.6	5	
	Feb. 1979	" "	"	1	0.009		<0.1	0.7	6.5	
	Aug. 1979	" "	"	1	0.115		<0.1	2.8		
	3/20-21/71	Larson, 1974	"	1-2		<0.01	0.17			DEQ Lab

Table A.1 Cont'd: Water Quality Data From Previous Studies

Lake	Sample Date	Source	Sample Site	Sample Depth (m)	TP (mg/L)	Phosphate PO_4-P (mg/L)	Nitrate NO_3-N (mg/L)	Chlor.-a (µg/L)	Secci Depth (m)	Comments
Clear cont'd	6/12-13/72	Larson, 1974	Atlas	1-2		<0.01	0.14		>6.1	DEQ Lab
	10/30-31/72	" "	"	"		0.07	0.22		4.0	" "
	8/10/60	Kruse & Oakley, 1961							3.7	from Larson, 1974
MUNSEL	6/23/92	Dagget, 1992	Atlas	JBS	0.01			1.43	6.32	DEQ Lab
	7/28/92	" "	"	"	<0.01			1.55	5.08	" "
	7/20/81	Johnson et al., 1985	"	"	0.014			0.9	4.7	" "
	5/1/81	" "	"	"	0.005			1.5	4.0	" "
	11/19/82	" "	"	"	0.017			2.4	4.8	" "
	Sept. 1978	Bryant et at., 1979	"	1			<0.01	10.1	4.5	
	Feb. 1979	" "	"	1		0.012	0.10	1.2	7.5	
	June 1979	" "	"	1		0.015	0.03	0.7	6	
	Aug. 1979	" "	"	1		0.092	<0.01	2.7		
	4/3-6/73	Larson, 1974	"	1-2	0.07	<0.01	0.12		4.6	DEQ Lab
	" "	" "	"	near bottom	0.07	0.01	0.14			
	6/11-12/73	" "	"	1-2	<0.03	0.01	0.05		4.0	"
	" "	" "	"	near bottom	0.03	<0.01	0.22			
	8/27-28/73	" "	"	1-2	0.03	<0.01	<0.01		5.0	"
	" "	" "	"	near bottom	0.03	<0.01	0.04			
	4/17-19/71	Kavanagh, 1973	"	0.5	0.0124			2		
	5/16-18/71	" "	"	"	0.0033			2.5		
	6/9-12/71	" "	"	"	0.0050			2.5		
	6/26-29/71	" "	"	"	0.0047			2.5		

Table A.1 Cont'd: Water Quality Data From Previous Studies

Lake	Sample Date	Source	Sample Site	Sample Depth (m)	TP (mg/L)	Phosphate PO_4-P (mg/L)	Nitrate NO_3-N (mg/L)	Chlor.-a (µg/L)	Secci Depth (m)	Comments
MUNSEL Cont'd	7/3/70	Larson, unpublished							5.5	from Larson, 1974
	11/16/67	Skeesick et al., 1970							4.0	" "
	12/20/67	" "							3.0	" "
	2/5/68	" "							5.5	" "
CLEAWOX	6/25/92	Dagget, 1992	Atlas	JBS	0.01			1.36	5.39	DEQ Lab
	7/30/92	" "	"	"	0.01			1.63	4.5	" "
	8/16/82	Johnson et al., 1985	"	"	0.005			0.9	5	" "
	4/3-6/73	Larson, 1974	"	1-2	<0.03	0.01	0.04			
	" "	" "	"	near bottom	<0.03	0.01	0.04			
	6/11-12/73	" "	"	1-2	<0.03	0.01	0.07			
	" "	" "	"	near bottom	<0.03	0.13	0.06			
	8/27-28/73	" "	"	1-2	0.03	<0.01	<0.01			
	" "	" "	"	near bottom	0.03	0.02	<0.01			
WOAHINK	6/24/92	Dagget,1992	"	JBS	<0.01			1.17	6.78	DEQ Lab
	7/29/92	" "	"	"	<0.01			2.02	4.88	" "
	7/7/89	Sweet, 1990	"	0.5	0.017	<0.002	0.04	2.8	5.9	
	9/24/89	" "	"	"	0.037	<0.003	0.08	3.8	4.9	
	7/19/89	Citizen Watch	"						3.8	
	9/27/89	" "	"						5.5	
	10/13/89	" "	"						4.4	
	7/28/81	Johnson et al., 1985	"	JBS	0.004			1.0	5.8	DEQ Lab
	5/2/82	" "	"	"	0.002			2.4	5.0	" "

Table A.1 Cont'd: Water Quality Data From Previous Studies

Lake	Sample Date	Source	Sample Site	Sample Depth (m)	TP (mg/L)	Phosphate PO_4-P (mg/L)	Nitrate NO_3-N (mg/L)	Chlor.-a (µg/L)	Secci Depth (m)	Comments
MOAHINK Cont'd	11/20/82	Johnson et al., 1985	Atlas	JBS	0.013			2.3	4.7	DEQ Lab
	Feb. 1979	Bryant et al., 1979	sta. 1	1		0.011	0.1	1.3	3	
	" "	" "	sta. 2	1		0.012	0.1	0.9	5	
	" "	" "	sta. 3	1		0.099	0.1	0.8	6	
CI	June 1979	" "	sta. 1	1		0.010	0.05	3.9	5	
	" "	" "	sta. 2	1		0.011	0.09	0.9	5	
	" "	" "	sta. 3	1		0.012	0.14	0.7	5	
	Aug. 1979	" "	sta.1	1		0.030	<0.1	0.8	5	
	" "	" "	sta. 2	1		0.041	0.03	1.5	5	
	" "	" "	sta. 3	1		0.038	0.03	1.2	5.5	
	6/12-13/72	Larson, 1974	sta. 4	1-2		0.03	0.09		5.5	DEQ Lab
	8/21-22/72	" "	" "	"		<0.01	0.07		7.3	" "
	10/30-31/72	" "	" "	"		<0.01	0.21		5.5	" "
	Oct. 1970	Maloney et al., 1975	?			0.001	0.024			
WC	7/2/69	Larson, 1970	?						6.6	from Larson, 1974
	8/16/60	McGie & Breuser, 1962							6.1	" "
	9/8/47	Saltzman, 1962							5.5	" "
SILTCOOS	7/28/81	Johnson et at., 1985	Atlas	JBS	0.018			8.4	1.8	DEQ Lab
	5/12/82	" "	"	"	0.069			3.7	1.9	" "
	11/20/82	" "	"	"	0.030			2.6		" "
	Feb. 1979	Bryant et al., 1979	"	1		0.038	1.0	7.9	2	
	" "	" "	"	4		0.034	0.9	8.9		

Table A.1 Cont'd: Water Quality Data From Previous Studies

Lake	Sample Date	Source	Sample Site	Sample Depth (m)	TP (mg/L)	Phosphate PO_4-P (mg/L)	Nitrate NO_3-N (mg/L)	Chlor.-a (µg/L)	Secci Depth (m)	Comments
SILTCOOS cont'd	June 1979	Bryant et al., 1979	Atlas	1		0.038	0.56	8.6	1.8	
	" "	" "	"	4		0.058	0.56	13.2		
	Aug. 1979	" "	"	1			0.02	16.3	1.3	
	" "	" "	"	4			0.02	27.3		
	1971	Kavanagh, 1973	"	0.5	0.019					annual mean TP
TAHKENITCH	5/3/82	Johnson et al.,1985	Atlas	JBS	0.012			6.6	2.5	DEQ Lab
	11/20/82	" "	"	"	0.022			2.1		" "
	4/3-6/73	Larson, 1974	"	1-2	<0.03	0.01	0.11			" "
	" "	" "	"	near bottom	0.03	0.02	0.13			" "
	6/11-12/73	" "	"	1-2	0.03	0.03	0.13			
	" "	" "	"	near bottom	<0.03	0.01	0.12			" "
	8/27-28/73	" "	"	1-2	<0.03	<0.01	0.04			" "
	" "	" "	"	near bottom	0.03	<0.01	0.04			" "
	Oct. 1970	Maloney et al., 1975	?	?		0.004	0.066			
EEL	6/25/92	Dagget, 1992	Atlas	JBS	0.02			4.06	4.19	DEQ Lab
	7/30/92	Dagget, 1992	"	"	0.01			4.45	4.06	" "
	8/4/81	Johnson et al., 1985	"	"	0.006			0.9	5.7	" "
N.TENMILE	5/5/82	" "	"	"	0.016			5.7	1.7	" "
	1/8/68	McHugh, 1972	?	JBS		0.02	0.12			
	" "	" "	?	near bottom		0.01	0.31			
TENMILE	7/8/89	Sweet, 1990	Atlas	1	0.040	0.006	0.04	11.9	2.0	
	9/24/89	" "	"	1	0.060	0.006	0.07	18.6	2.7	

Table A.1 Cont'd: Water Quality Data From Previous Studies

Lake	Sample Date	Source	Sample Site	Sample Depth (m)	TP (mg/L)	Phosphate PO_4-P (mg/L)	Nitrate NO_3-N (mg/L)	Chlor.-*a* (µg/L)	Secci Depth (m)	Comments
TENMILE cont'd	5/5/82	Johnson et al., 1985	Atlas	JBS	0.013			2.7	3	DEQ Lab
	11/22/82	" "	"	"	0.013			6.6	2.5	" "
	4/30/69	McHugh, 1972	?	JBS		0.007	0.36			
	1/8/68	" "	?	"		0.017	0.31			
	Oct. 1970	Maloney et al., 1975	?	?		0.001	0.004			
LOON	9/18/82	Johnson et al., 1985	Atlas	JBS	0.004			0.5	6.5	DEQ Lab
	8/12-13/75	DEQ	"	"	0.03	0.003	<0.01			" "
FLORAS	8/3/81	Johnson et al., 1985	"	"	0.008			0.7	2.6	" "
	8/12-13/75	DEQ	"	"	0.03	0.003	<0.01			" "
	4/30/69	McHugh, 1972	?	JBS		0.01	0.14			
GARRISON	6/26/92	Dagget, 1992	Atlas	JBS	0.03			8.17	2.46	DEQ Lab
	7/31/92	" "	"	"	0.04			20.08	2.26	" "
	1988-89	SRI, 1990	Atlas	JBS	0.035			6.3	3.9	annual mean
	" "	" "	site 2	JBS	0.074			14.5	13.5	" "
	1985	DEQ	Atlas	"	0.035			9.6	1.9	annual mean; from SRI
	"	"	site 2	"	0.092			19.0	1.4	" "
	11/22/82	Johnson et al., 1985	Atlas	"	0.027			2.7	4.5	DEQ Lab
	5/4/82	" "	"	"	0.012			3.3	1.6	" "
	8/3/81	" "	"	"	0.103			27.2		" "

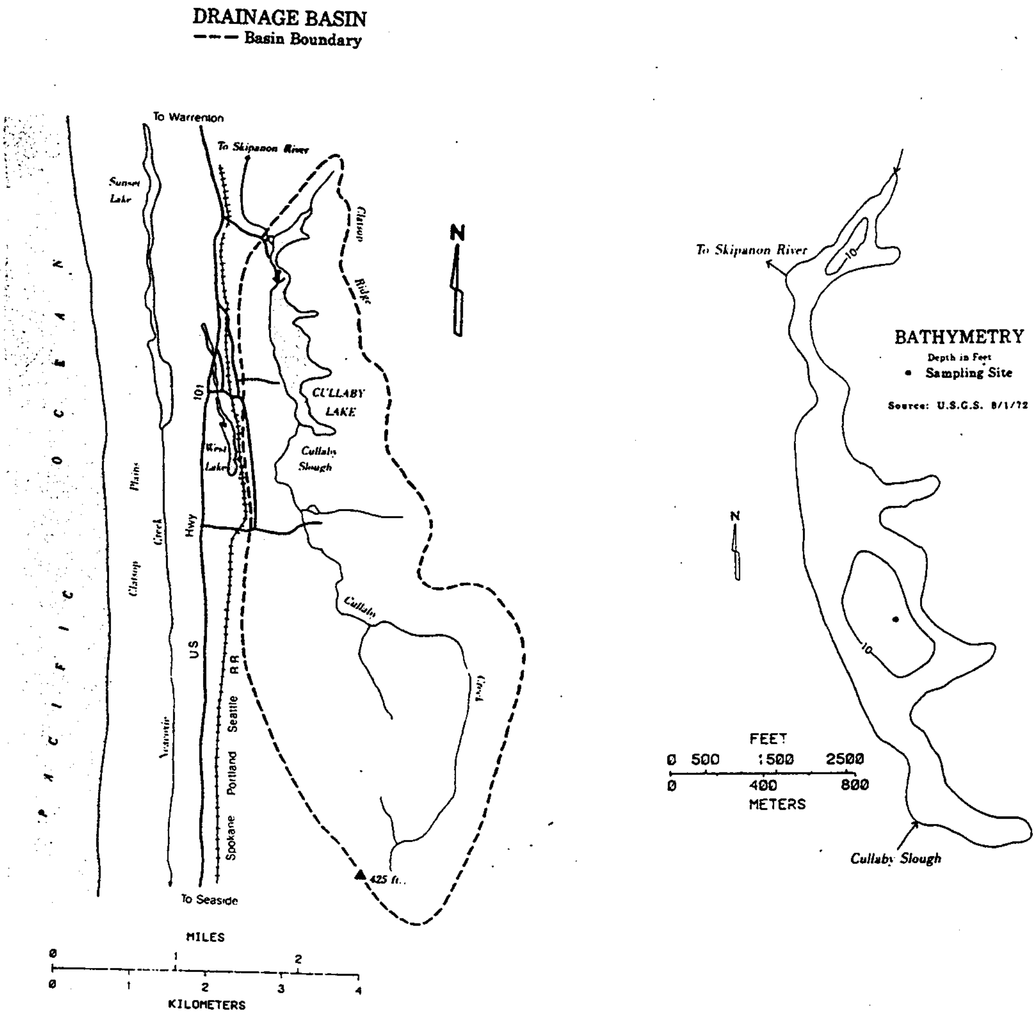


Figure A.1: Cullaby Lake

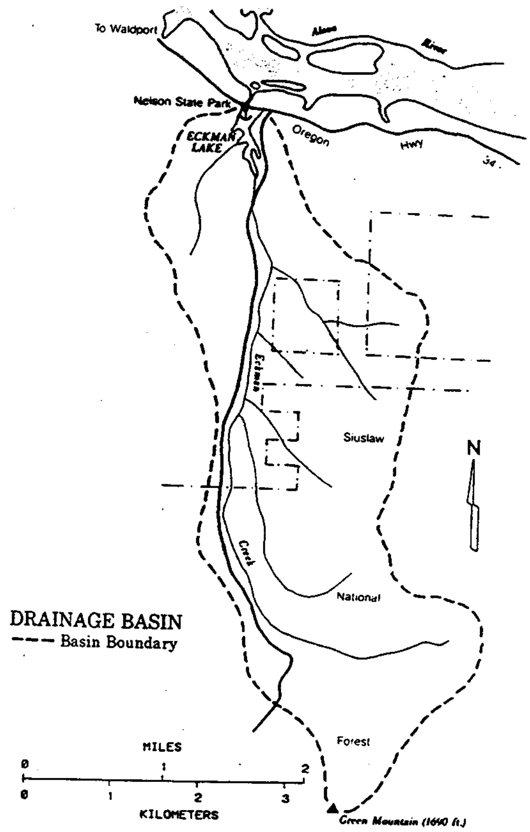
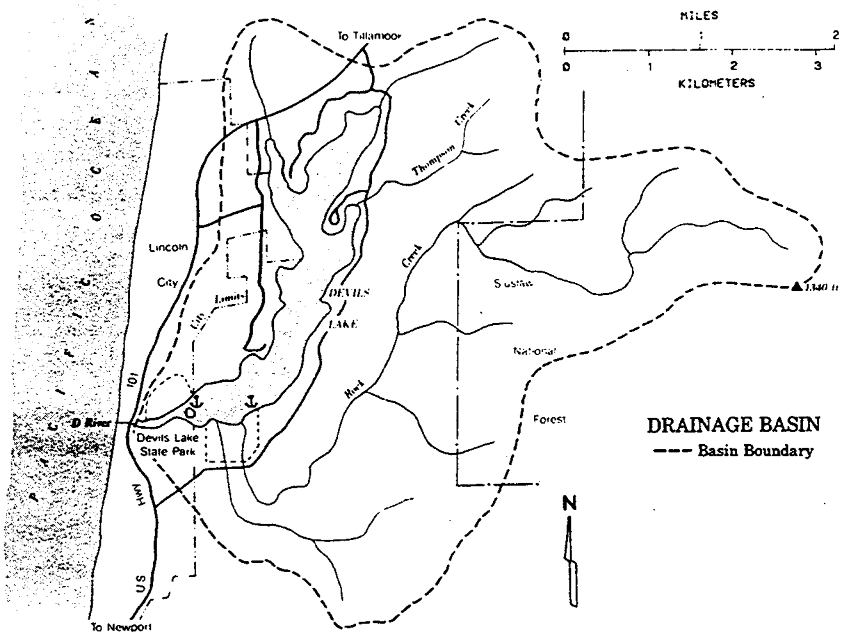


Figure A.2: Eckman Lake



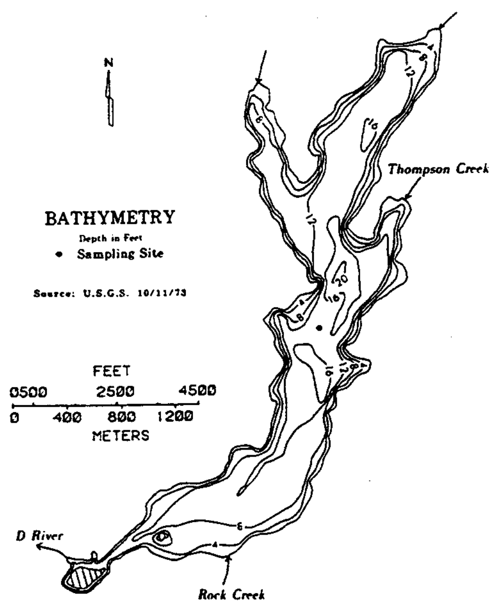


Figure A.3: Devils Lake

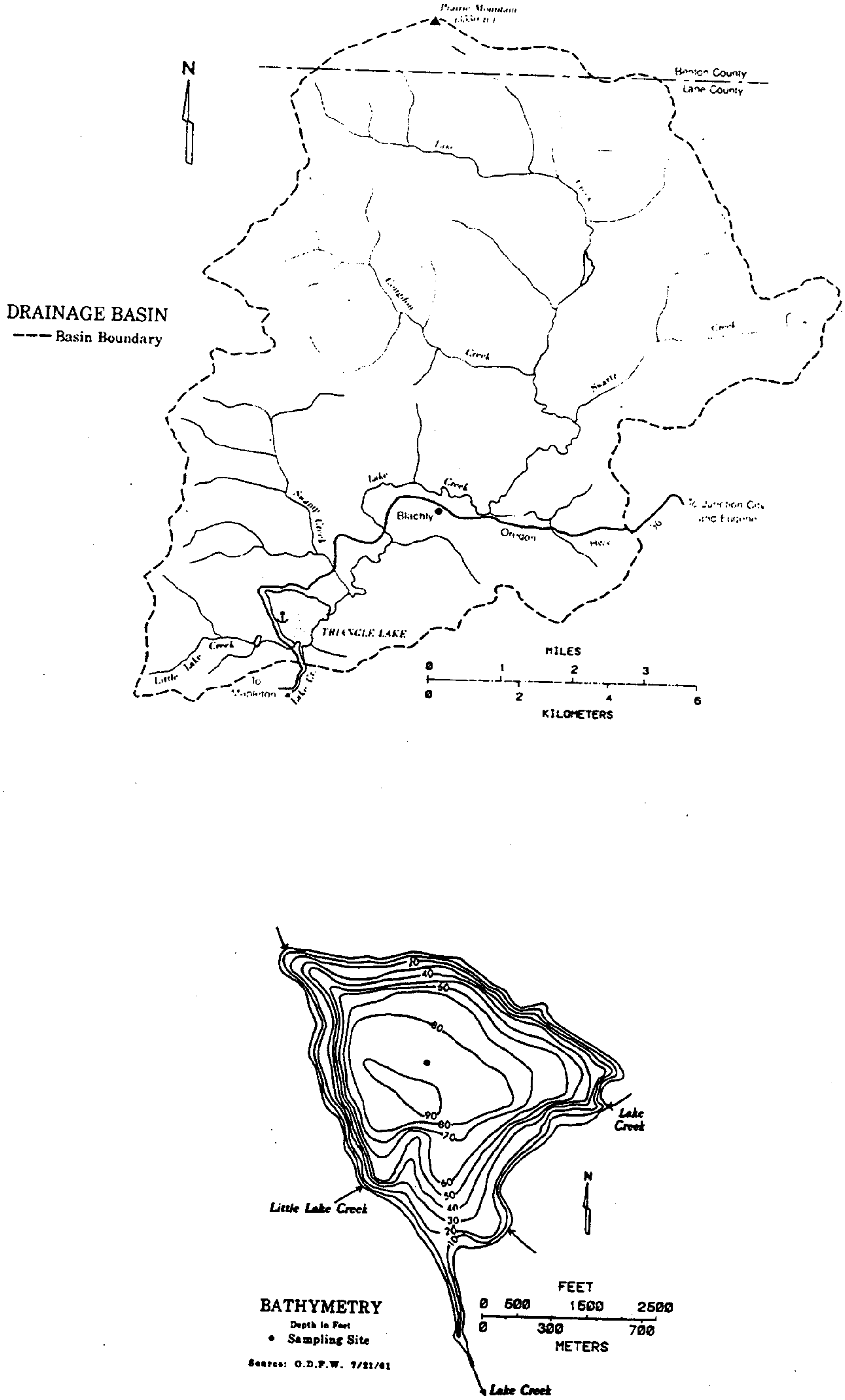
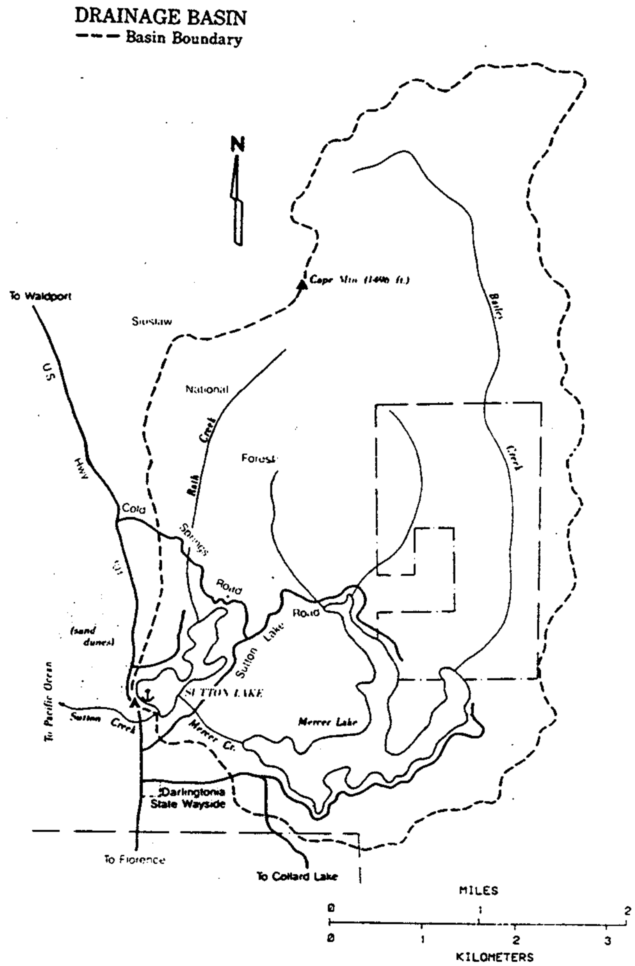


Figure A.4: Triangle Lake



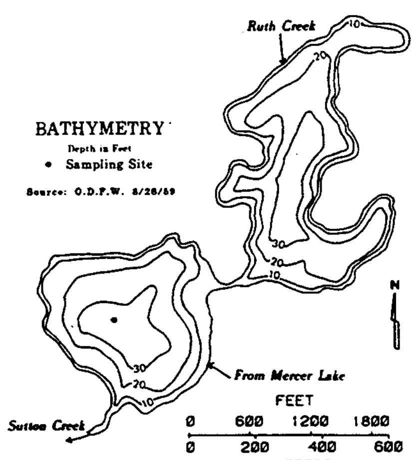
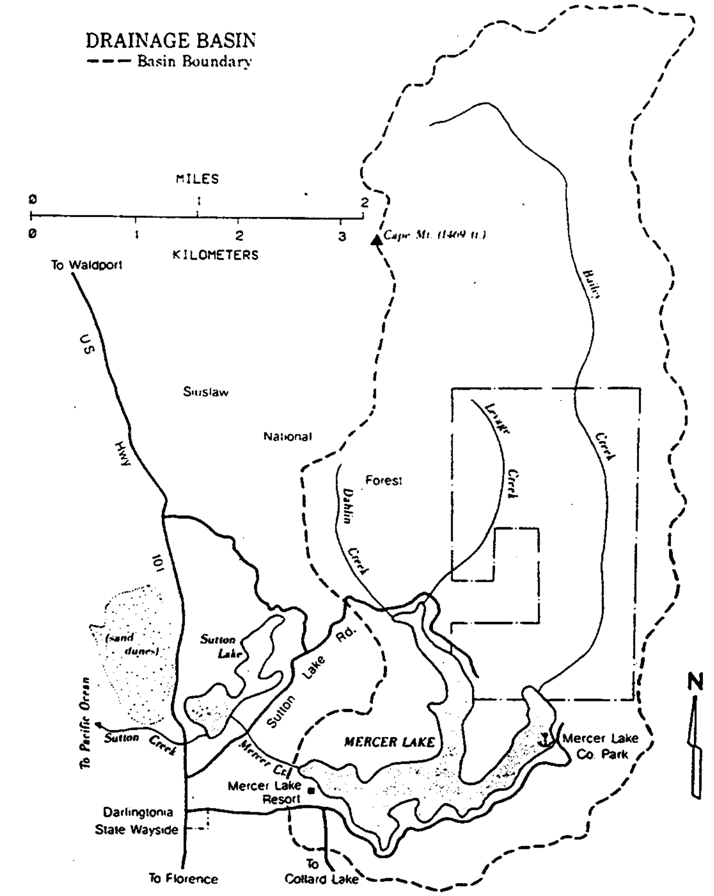


Figure A.5: Sutton Lake



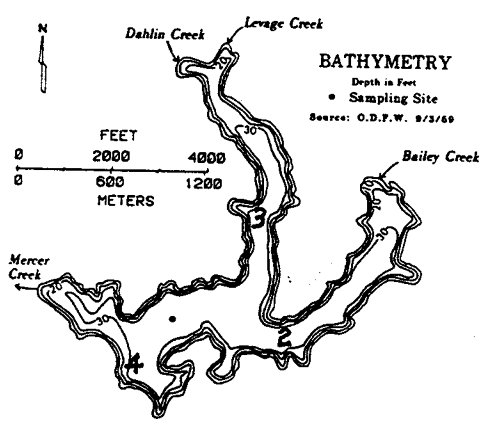
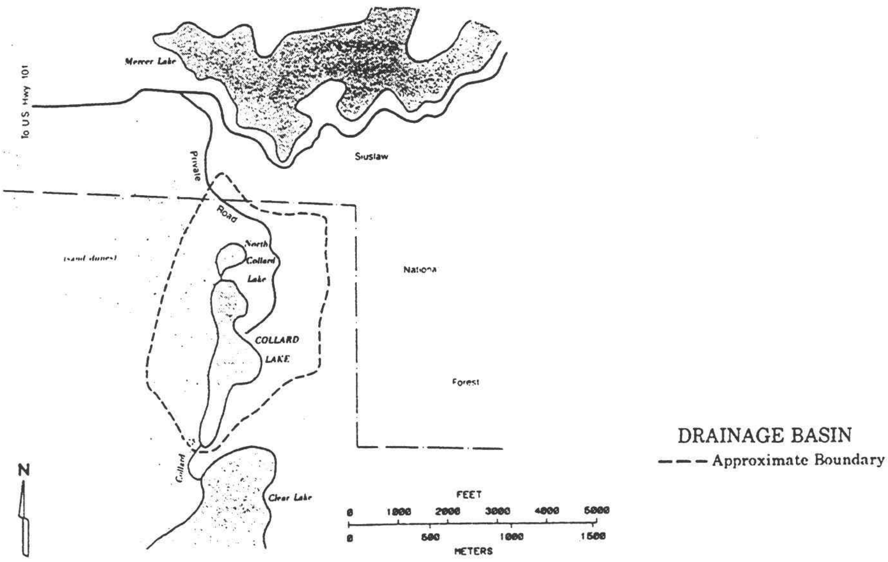


Figure A.6: Mercer Lake



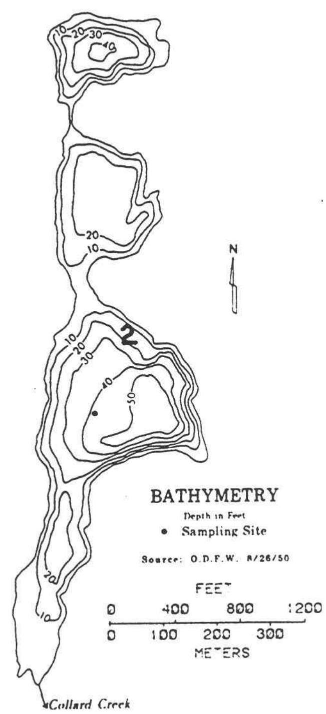
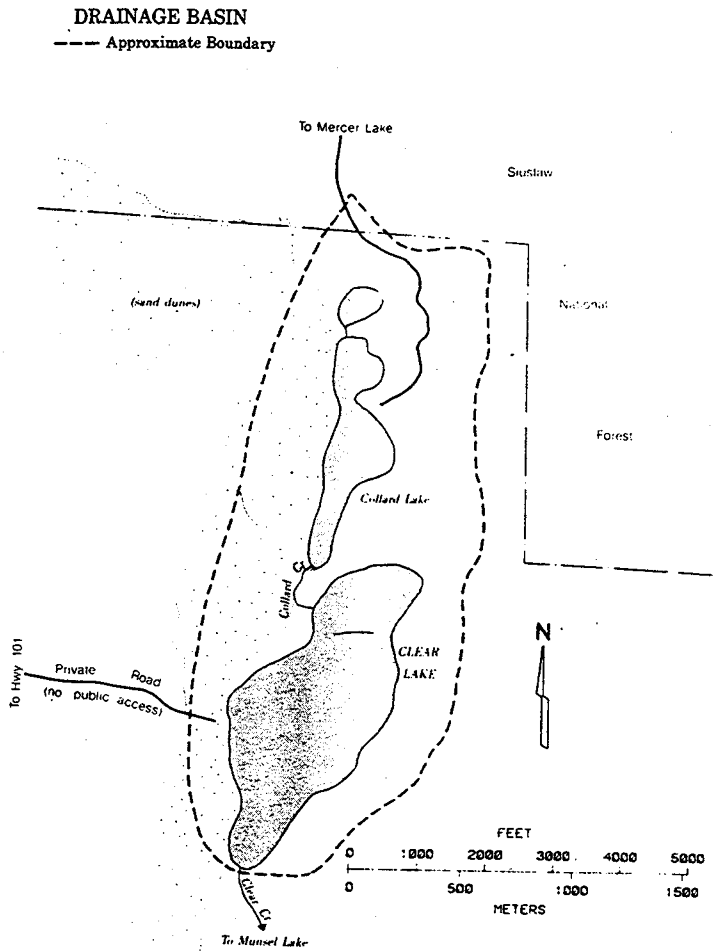


Figure A.7: Collard Lake



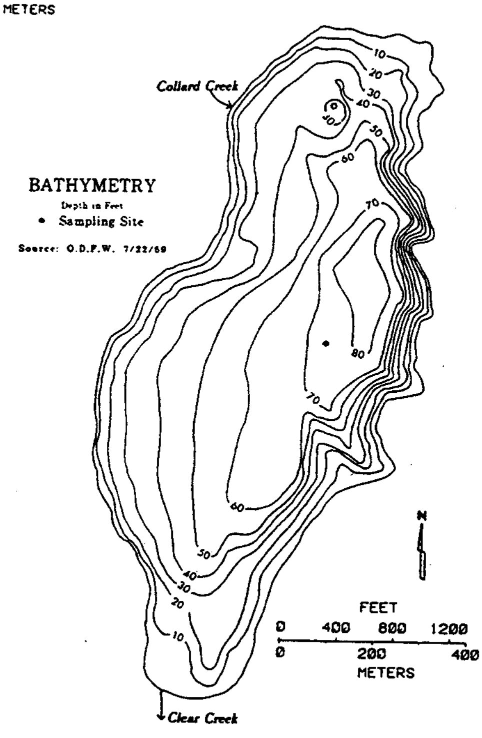
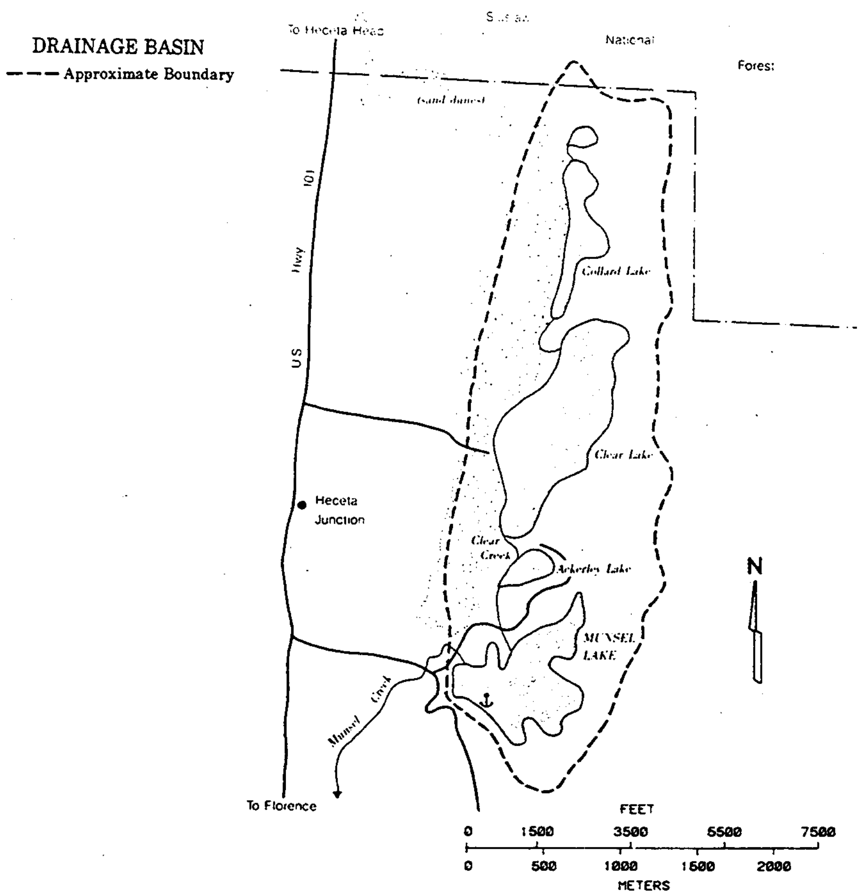


Figure A.8: Clear Lake



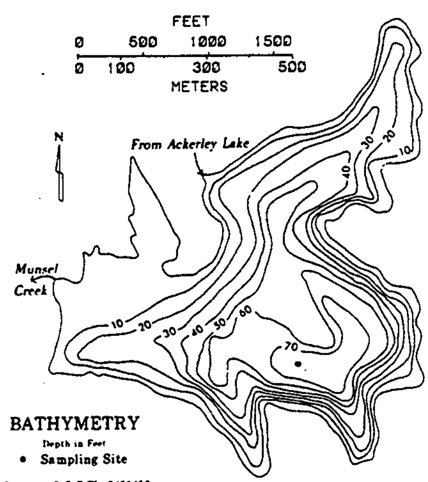
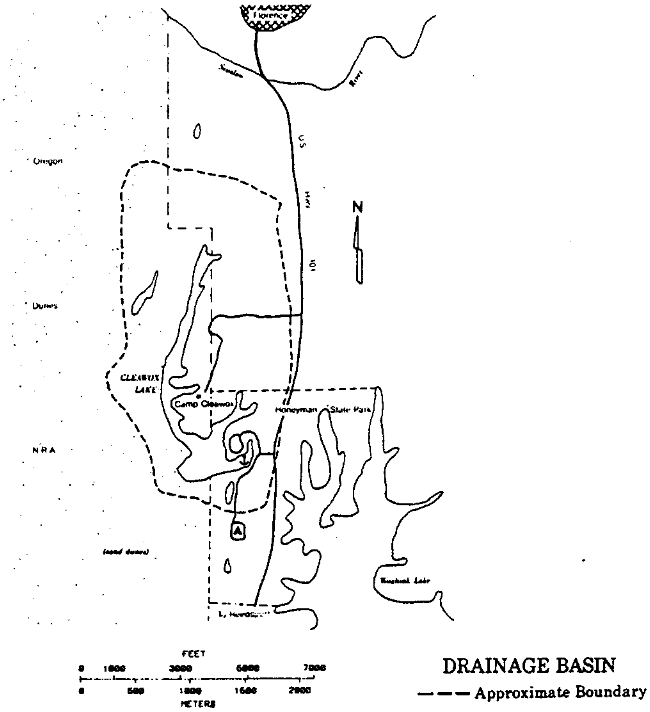


Figure A.9: Munsel Lake



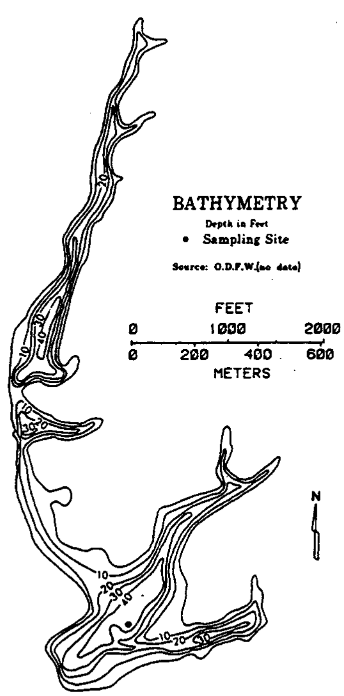


Figure A.10: Cleawox Lake

DRAINAGE BASIN
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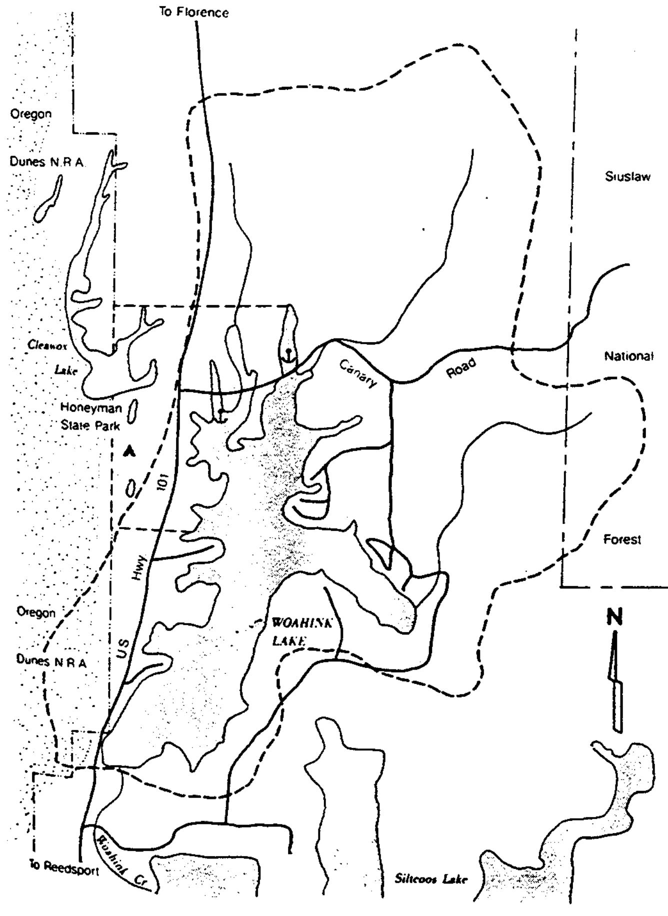


BATHYMETRY
Depth in Feet
• Sampling Site
Source: O.D.F.W. 2/24/61

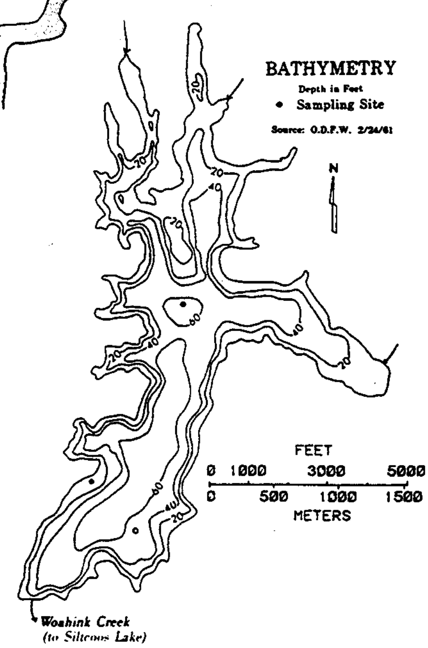
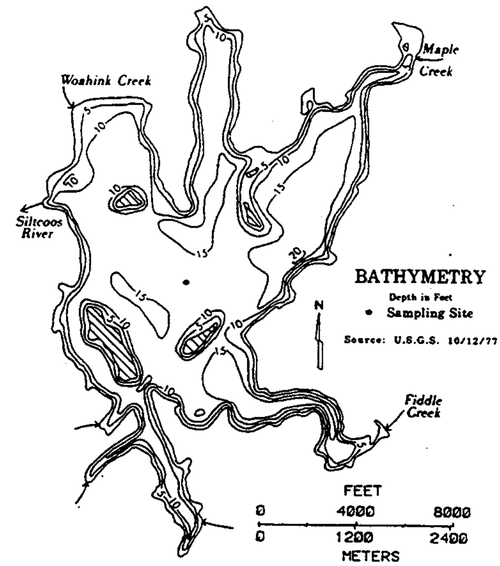


Figure A.11: Woahink Lake



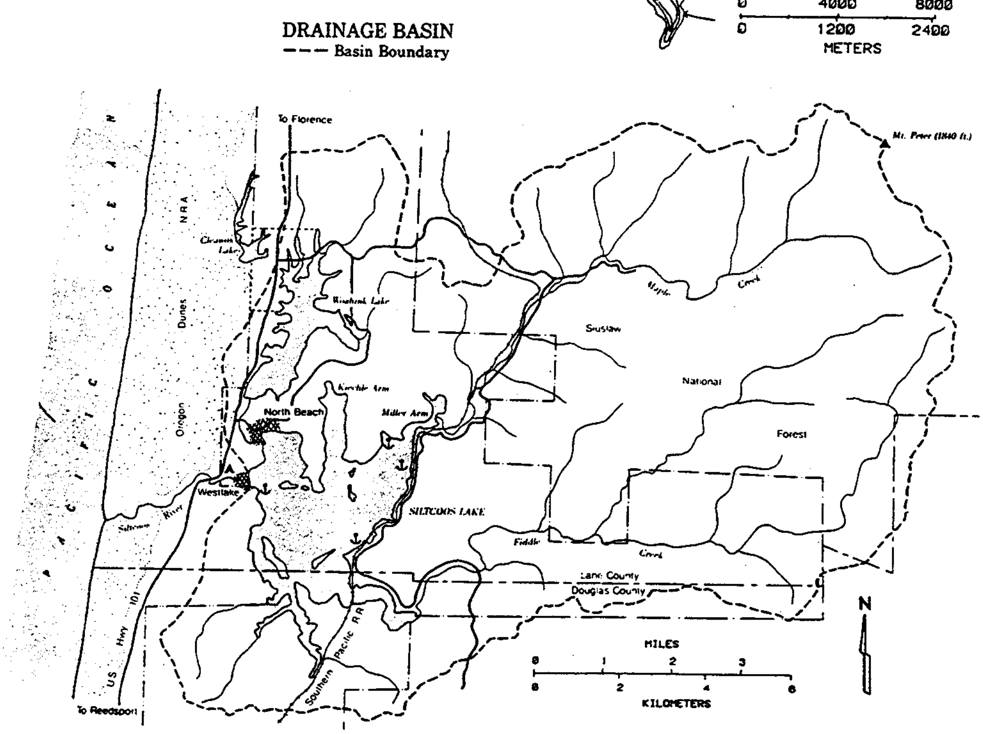
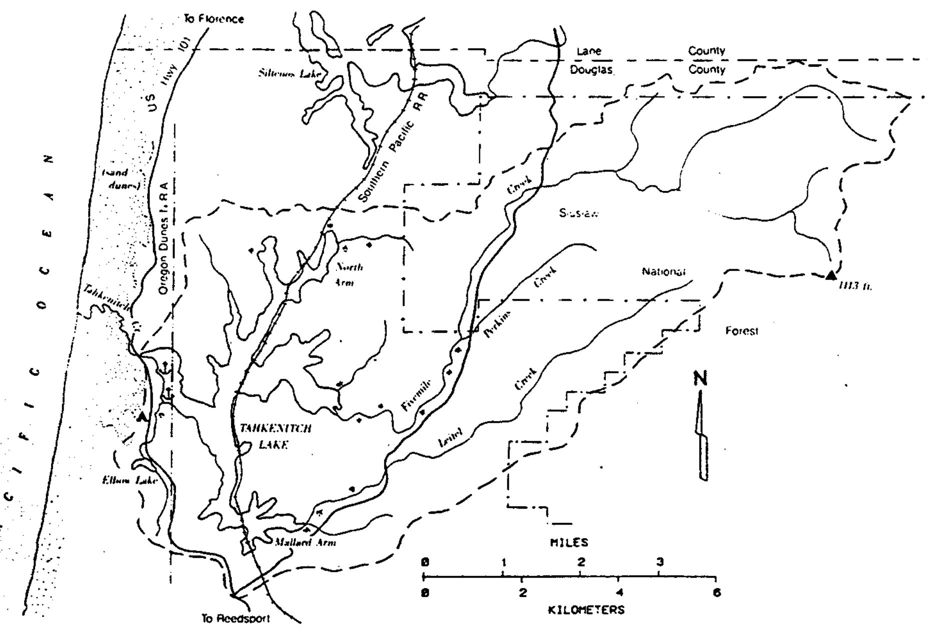


Figure A.12: Siltcoos Lake



DRAINAGE BASIN
--- Basin Boundary

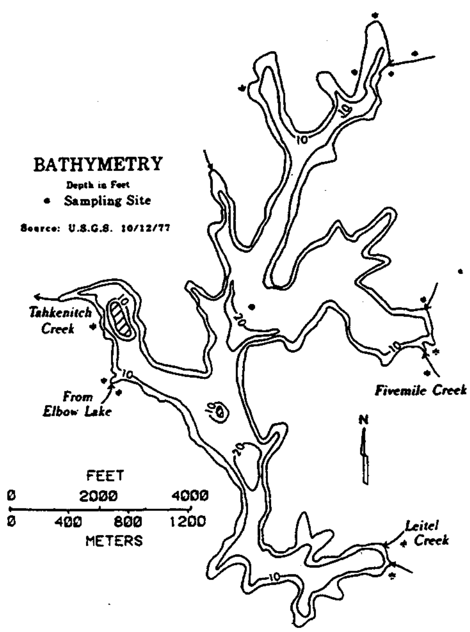
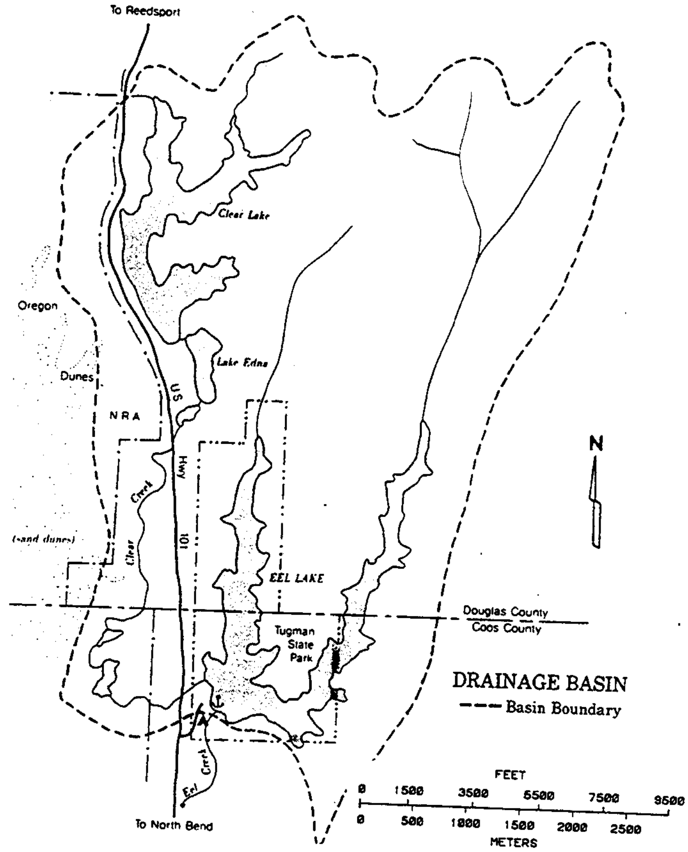


Figure A.13: Tahkenitch Lake



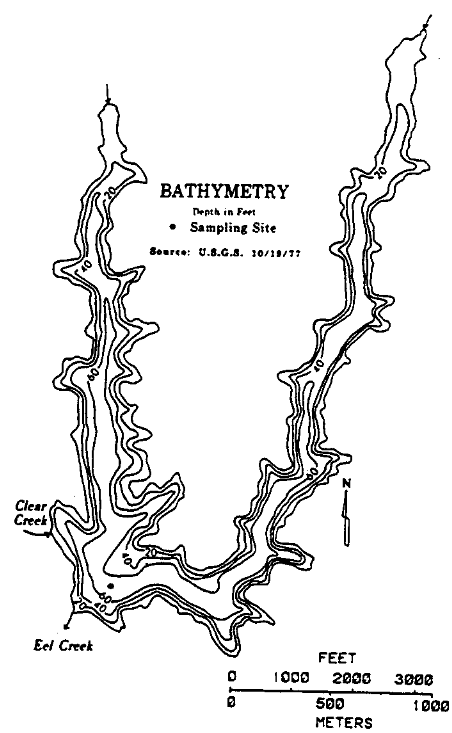
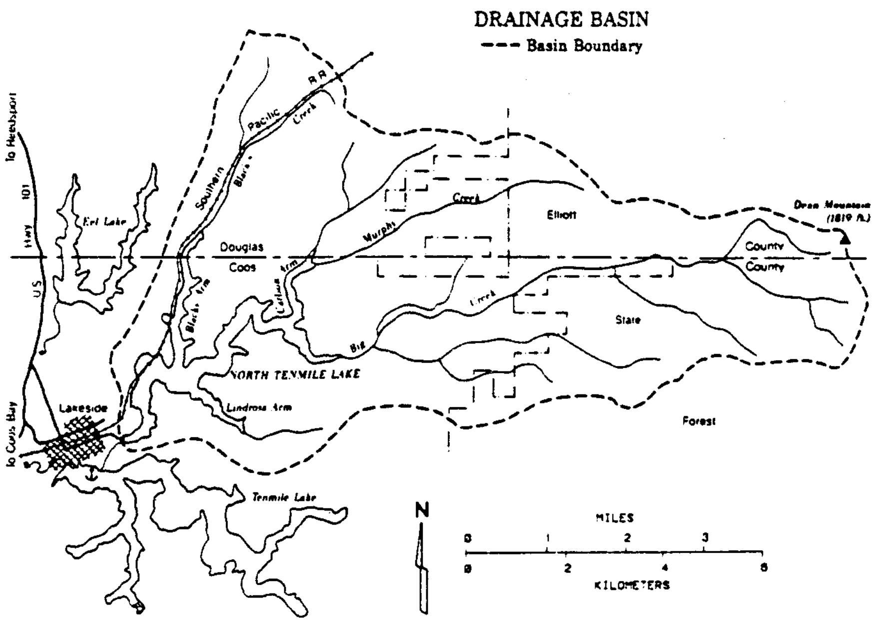


Figure A.14: Eel Lake



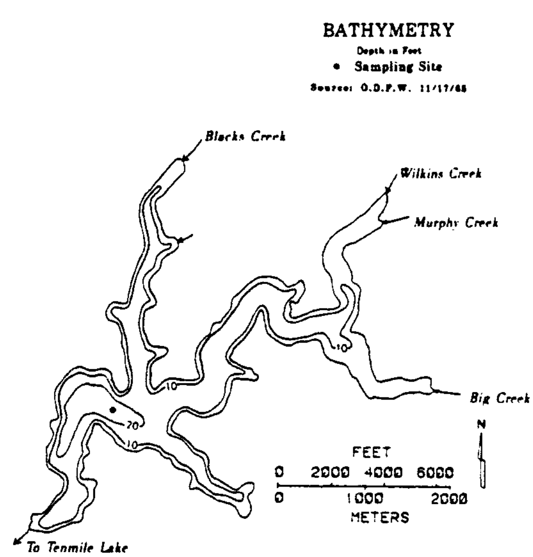
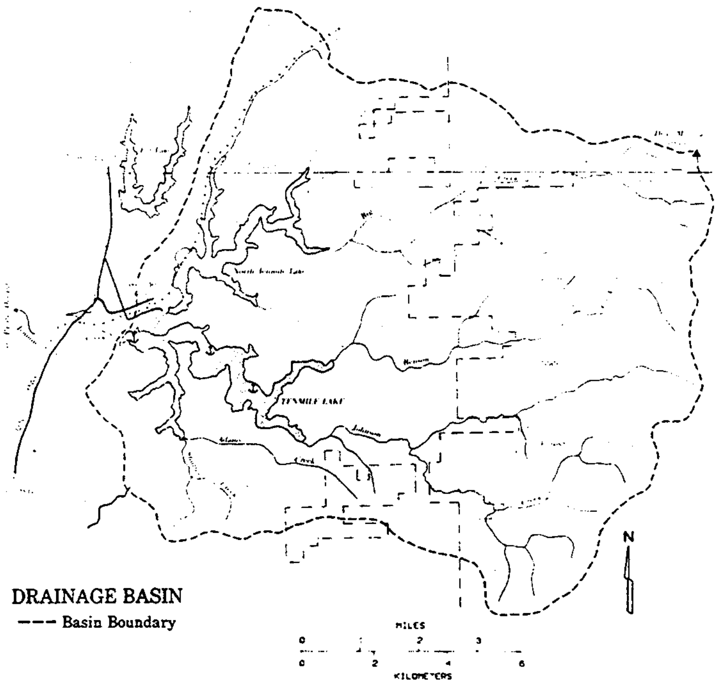


Figure A.15: North Tenmile Lake



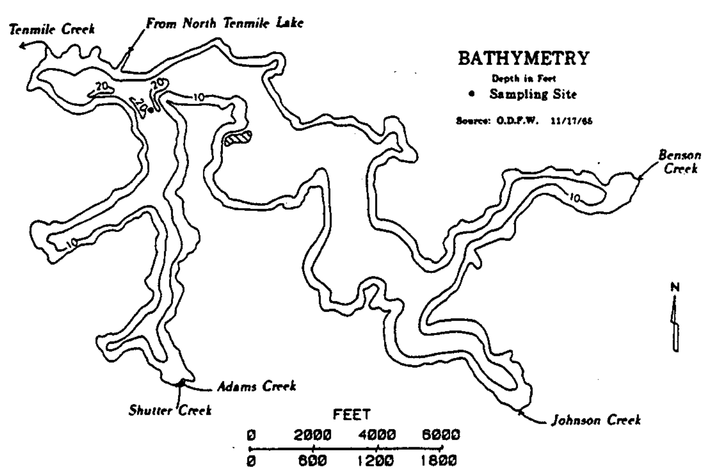
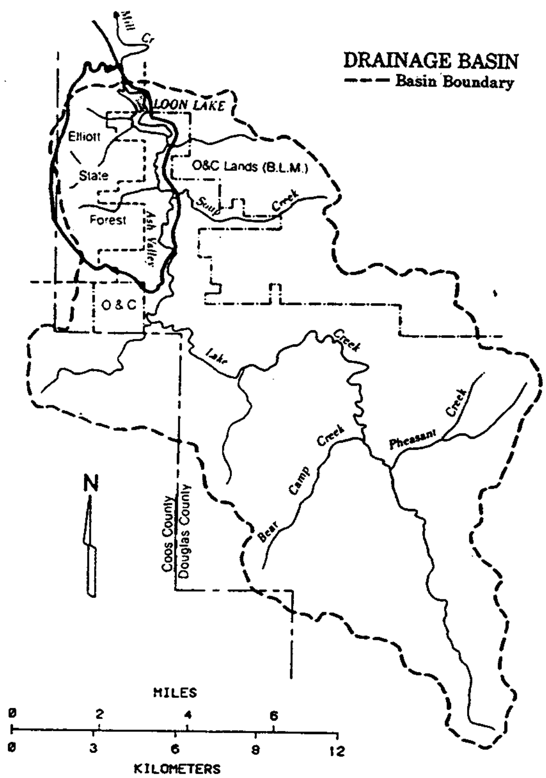


Figure A.16: Tenmile Lake



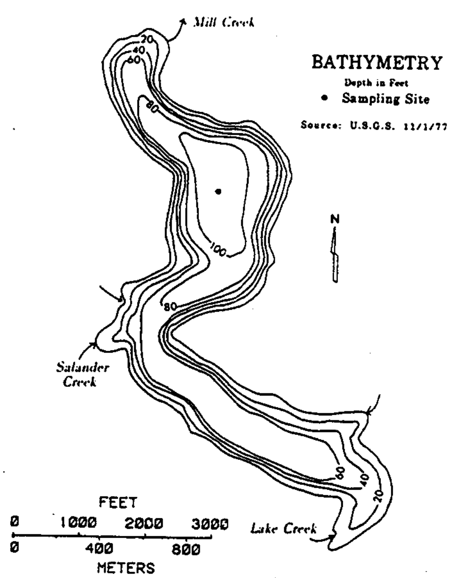


Figure A.17: Loon Lake

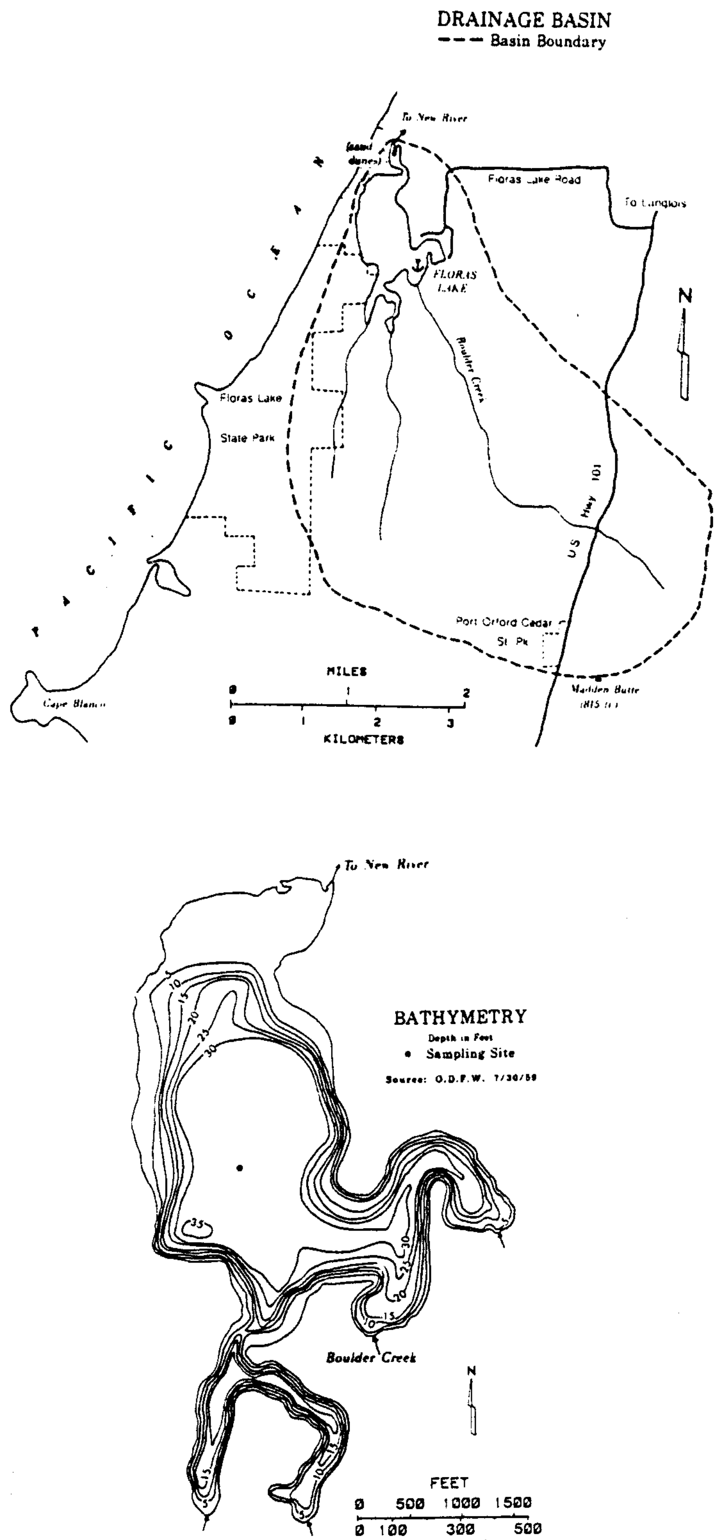


Figure A.18: Floras Lake

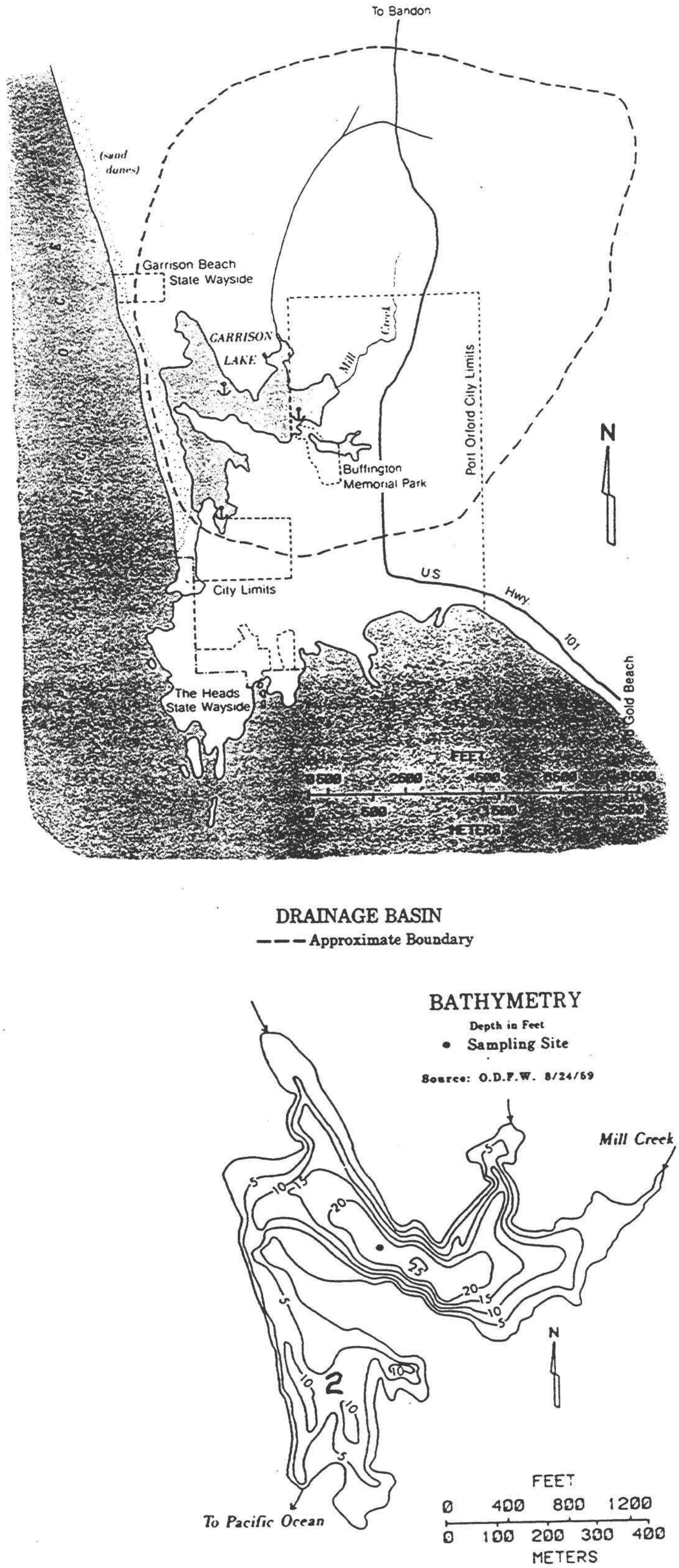


Figure A.19: Garrison Lake